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Revitalizing Brownfields and Greyfields at Oak Ridge National Laboratory to Promote Urban Forestry Management

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Revitalizing Brownfields and Greyfields at Oak Ridge National Laboratory to Promote Urban Forestry Management

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ABSTRACT

The Oak Ridge National Laboratory (ORNL), formally known as X-10 or Clinton Laboratory, was established during the early 1940s to house the world's first nuclear reactor. The site was originally used for the production and separation of plutonium during World War II. Today, the ORNL site is used for multiple purposes including research facilities and utility infrastructure, to meet the national goals and objectives of the Department of Energy. Activities associated with its historical and contemporary use has led to severe land disturbance along with excessive inputs of toxic chemical waste. Many issues that impact the ORNL campus and the surrounding forest land-use change and development, land erosion, soil contamination, and compaction, altered vegetation, forest pest, and invasive plants. A study was conducted to (1) investigate trees species diversity, determine diameter at breast height (DBH) distribution, evaluate tree health, and to quantify ecosystem services and values associated with landscape trees. (2) chemical soil composition within managed vegetation sites on the ORNL campus. There were a total of 1160 trees, composed of 62 species, and 30 genera. The species with a high relative abundance are *Acer rubrum* (10.7%) and *Cercis canadensis* (9.6%). The most important species in terms of percent population, leaf area size, and structural value are *Acer rubrum* (19.3), *Quercus palustris* (17.4), *Juniperus virginiana* (15.0), *Pinus strobus* (11.2), and *Quercus phellos* (7.1). Basic soil properties, such as pH and total element content were characterized. The concentrations of twenty-one elements were determined: Al, As, Ba, Ca, Co, Cd, Cr, Cu, Fe, K, Li, Mg, Mn, Mo, Na, Ni, P, Pb, Sr, and Zn. The elemental concentrations in soils from the ORNL campus were compared to those of native soil profiles of the eastern Tennessee region and median levels for uncontaminated world soils. There were significant correlations between elements Al, Cr, Fe, K, Li, Ni, Pb, and Sr. Results show that elemental concentrations in soil samples from the ORNL site

are within the ranges tabulated for soil profiles of the eastern Tennessee region, suggesting that metal contamination has not occurred.

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INTRODUCTION

Introduction

Many cities in the United States today owe their land use growth and expansion to industrialization. American industrialization during the 19th and 20th century is largely responsible for much of the urbanization and urban development in the United States (Kim, 2005). Urbanization refers to the redistribution of populations from rural or farmland to urban towns and cities. Urban land development typically includes clearing of forested or rural land, disturbance, and removal of soil, and building of infrastructure (Pouyat *et al*, 2007). Urbanization in America saw the development of many new towns and cities that continued to grow as more people were attracted by employment opportunities. As the U.S. economy shifted from agriculture to manufacturing, cities expanded their functions from rural areas to the location of industrial activity (Michaels *et al.*, 2012).

Prior to industrialization, 5% of Americans lived in urban areas (Rees, 2016). As industrialization expanded, by 1890 the population residing in urban areas increased by 30%, mostly in the northern cities such as New York, Philadelphia, and Chicago. The late 19th century brought advancements in transportation that included highways, streetcars, trolleys, and railroads, (Rees, 2016). Migration to cities was encouraged by the opportunity for employment, along with improvements in transportation and housing construction. Urban expansion continued to expand during the World War II period (1939-1945). Pre-and Post WWII development led to the urbanization of the southern half of the country, particularly in Texas, Arizona and southeastern states. By the 1990s, 75 % of the U.S. population lived in an urban setting (Rees, 2016; Boustan *et al.*, 2013).

In the year 2012, approximately 81 % of the U.S. population lived in urban areas (U.S. Census Bureau, 2012). In one decade dating 1990-2000, urban areas in the U.S. increased by 13% (Lubowski *et al.*, 2006). In a three-decade span dating 1960-1990, more than 100 million acres of land in the U.S. were claimed by urban expansion (Heimlich and Anderson, 2001). By the year 2000, approximately 3.5 % of land in the U.S. was classified as urban with 25 % of the land being functionally tied to urban areas, which are considered metropolitan areas (Dwyer and Nowak, 2000). Urban areas are expanding and consequently, forestland surrounding these areas are altered or destroyed. Nowak and Greenfield (2012), estimated that in the U.S. 4.0 million trees per year of forest cover has been destroyed. Urban expansion has transformed the natural landscape through anthropogenic fragmentation of forest and natural habitats, development of impermeable surfaces, thus inducing strong pressures over environmental systems (Weber, 2013).

Defining Urban Forest and Urban Forestry

Urban forests are defined as "*the sum of all woody and associated vegetation in and around dense human settlements, ranging from small communities in rural settings to metropolitan areas*" (Miller *et al.*, 2015). The U.S. Forest Service and many urban forestry organizations describe urban forests as "*all publicly and privately-owned trees within an urban area including individual trees along streets and in backyards, as well as stands of the remnant forest*" (Nowak *et al.*, 2010). The concept of urban forestry emerged in the mid-1960s, which referred specifically to the management of trees in an urban area (Johnston, 1996). Urban forestry is a specialized branch of forestry and is defined as "*the art, science, and technology of managing trees and forest resources for their present and potential contribution to the physiological, sociological and economic well-being of urban society*" (Konijnendijk *et al.*, 2006). Over the past four decades, urban forestry has garnered international recognition from a diverse community of practitioners, researchers, and governments (Miller *et al.*, 2015; Nail, 2008). This recognition has resulted in the expansion in both the science and management of urban forests (Seamans, 2013). With expanding human populations in urban areas, preserving the health and intrinsic value of urban forests has become a national and international goal.

Urban forests are typically comprised of three different types of urban landscapes, native or natural, functional or managed, and adaptive (ruderal) landscapes. Natural or native landscapes consist of fragmented natural woodlands and wetlands that existed before it is acted upon by human culture. These landscapes are comprised of native vegetation on relatively undisturbed soils that require minimal to moderate maintenance. Functional landscapes are intensively managed areas such as commercial properties, residential lawns, parks, street trees, mostly comprised of cultivated plants on manufactured soils. Adaptive or ruderal landscapes sometimes referred to as greyfields, are post-industrial land, vacant lots, infrastructure, degraded wetlands, and woodlands. Generally, these landscapes are undermanaged, contain compacted or fill soils, and are dominated by spontaneous vegetation (Tredici, 2010).

Urban forests are an integral part of urban environments, which are composed of green and grey infrastructure that interacts significantly to affect the quality of life for residents that live in these environments. The urban forest significantly influences human physical and mental health providing places for people to walk, bike, sit and explore their community (Nowak and Dwyer, 2007). Tyrvainen et al., 2005 studied the benefits, values, and uses of how cities and towns utilize their green infrastructure. The study looked at four benefit categories social, aesthetic/architectural, climatic/physical, and economic/ecological benefits. Specifically, aesthetics/architectural benefits create natural barriers and serve as screens and visual filters for unsightly and sometimes necessary urban activities. Climate/physical, impact urban climates through temperature, humidity control, air pollution reduction, and erosion control. Economic/ecological/ social provide an abundance of recreation sites, attract diverse wildlife, and increases property values in urban communities. These four benefit categories are urban forest community valuable assets that require ongoing care and stewardship.

Urban Forest Ecosystem Services

Ecosystem services are "*the benefits human populations derive, directly or indirectly, from ecosystem functions"* (Weber and Medhi, 2012; Costanza *et al.*, 2014). Ecosystem services are typically grouped into four categories: provisioning, regulating, supporting, and cultural services (esa.org). Ecosystem services generated by urban ecosystems have crucial importance for the quality of life and public health for urban residents. Specifically, in urban areas these services direct maintenance processes, natural resources, and influence cultural services (Weber, 2013).

Many of the ecosystem services provided by urban forest are correlated with tree density and canopy health. Urban Tree Canopy (UTC) assessment is a measurement of the extent of the urban forest and the amount of potential ecosystem services provided by vegetation in an urban environment (Nowak and Greenfield, 2012). Urban forest research has taken a special interest in the role of urban vegetation in air pollutants abatement because it is a global environmental problem impacting most major cities worldwide (Ning *et al.*, 2016). Maintaining large canopy trees is key to air pollution abatement because large trees have the capacity to remove 60 to 70 times more pollution than small trees because of their leaf surface (Diduck, 2013). Therefore, sustaining the health and longevity of mature trees is essential to maximizing air quality benefits. In addition, by planting tolerant trees species in areas where air pollution concentrations are high may enhance air quality benefits (McPherson, 1998). Trees intercept rainwater and divert it into the soil where bacteria and microorganisms filtrate pollutants. Rainfall interception is an important service provided by urban vegetation that reduces stormwater runoff and the amount of sediment, organic matter, and pollutant that reach waterways. Assessing ecosystem services that contribute to the quality of life in urban environments in quantitative and economic terms can help to direct proper management and stewardship of urban vegetation (Clark *et al.*, 1997).

Overview Economic Valuation of Ecosystem Services in the U.S.

In the U.S., urban forests support 79% population providing ecosystem services to more than 220 million people (Nowak and Greenfield, 2010). A 2002 study of urban trees in the U.S., estimated that there were approximately 3.8 billion trees that constitute U.S. urban forests with a structural value of $2.4 trillion (Nowak *et al.*, 2002). Trees cleanse the air by absorbing carbon dioxide, sulfur dioxide, nitrous oxides, exhaust from vehicles, thus reducing ozone emissions (McPherson *et al.,* 2006). According to 2002 evaluation of carbon benefits in the U.S., urban trees were estimated to store 700 million tons of carbon and sequester carbon at a rate of 22.8 million tons per year with a combined economic value of $14.5 billion (Nowak and Crane, 2002). Research has compared the uptake of carbon in urban trees and forest trees and concluded that a single urban tree can contain about four times more carbon than a single tree in natural forest (Nowak and Crane, 2002). Trees and vegetation provide cooling effects by lowering surface and air temperatures through evapotranspiration and shading. Trees reduce urban heat island temperatures by 10-20 ° F, leading to reduced ozone levels and improvements in air quality standards (Alliance for Community Trees, 2011). Studies have shown how, trees in Berkeley, CA, on an annual basis produce environmental benefits with a monetary value of $3.25 million (McPherson, 2005). While, trees in Mecklenburg Country, NC, provide stormwater management and air pollution abatement benefits valued at $200 million per year (American Forests, and U.S. Forest Service, 2010). Street trees in New York City were estimated to intercept 890 million gallons of stormwater annually with a monetary value of over $35 million (Nowak *et al.,* 2002). Additionally, studies have found that the presence of trees and vegetation on a residential property can increase property values up to 37 % (Foster *et al.,* 2011).

Adverse Effects of Urban Environments

Urban areas are often aggressive environments, accompanied by many challenges and threats, including adverse climatic conditions, impervious surfaces, limited growing space, and atmospheric pollution (Konijnendijk *et al.*, 2006). Impervious surfaces play a vital role in urban landscapes but are an environmental concern. These surfaces facilitate transportation and provide shelter, however; increases in impervious surfaces through the development of infrastructures modify urban air and water resources (Chithra *et al.,* 2015). The increases in infrastructure amplify local temperatures and create urban heat islands (UHI) which is one of the most prominent issues associated with urbanization and industrialization. The UHI effect describes the differences between urban and rural temperatures, which is a result of climate modification and changes to the form and composition of the land surface and atmosphere when vegetation is replaced by asphalt and concrete (U.S. EPA, 2017). For example, Tucson, Arizona experienced rapid urban development from 1949 to 2005, the average annual minimum temperatures increased by 5.4 ° F, with 3.6 ° F of this increase attributed to the UHI effect (Brazel *et al.,* 2007). Phoenix, Arizona experienced a similar increase in temperature during the same period with nighttime minimum temperature increasing by approximately 9 ° F and the daily average by 5.5 ° F. The UHI effect consequently affects ozone production, pollution emissions, building energy use, and ultimately influences human health and comfort (U.S. EPA, 2017).

Trees play a vital role throughout the hydrological cycle. Impervious surfaces and the absence of tree canopy cover can significantly affect hydrology processes in urban environments (Berland *et al.,* 2017). Leaves and branches intercept rainfall preventing it from reaching the ground and becoming surface runoff and aids in the evaporation process by capturing water (Kuehler *et al.,* 2017). Urban development has altered the natural hydrologic processes by removing forest cover, replacing the vegetative ground cover with artificial lawns, covering the soil with impermeable surfaces, which leads to increased stormwater runoff and reduced water quality (Tenneson, 2014). Therefore, restoring or maintaining the tree canopy in urban ecosystems may help with stormwater management (Kuehler *et al.,* 2017). Studies quantifying rainfall interception by tree canopy in urban areas are limited, however, multiple studies assessing tree canopy in natural forests are documented (Sun and Lockaby, 2012). Xiao *et al.*, (2000) explains that tree canopy structure differs in natural forest and open-grown trees urban settings in terms of tree spacing, leaf area, leaf surface characteristics, leaf angle distribution. Chithra et al., (2015) suggest that with adequate growing conditions, trees in urban settings can provide immense stormwater benefits useful to managers and engineers as they work to mitigate the effects of stormwater runoff.

<u>Natural vs Urban Soils</u>

Natural forest soils are comprised of the original geologic material deposited across the topography of the landscape, acted upon by various abiotic and biotic factors, and weathered over time by the climate conditions of the region. Hence, soil formation is a long-term process (Brady 1984). Diverse soils are formed in different localities due to the diversity of abiotic and biotic factors over the landscape. Forest soils are subjected to fewer disturbances than urban soils. Natural and urban soils are similar because they are influenced by the same abiotic and biotic interactions, however, urban soils are formed through the process of urbanization. Urban soils undergo a unique process and therefore cannot be separated from the geographic constraints of the process (Craul, 1985). There a several definitions for urban soils proposed in literature: (1) "*urban* *soil is a material having a non-agricultural, anthropogenic surface layer more than 50 cm thick that has been produced by mixing, filling, or by contamination of land surfaces in urban and suburban areas*" (Bockheim 1974); (2) "*urban soil as a material that has been manipulated, disturbed or transported by human activities in the urban environment and is used as a medium for plant growth and to support human activities*" (Craul, 1999). Some essential functions such as carbon storage, nutrient cycling and transformation, and pollution interception from human activities take place in urban soils (Cunningham *et al.,* 2008).

Urban soils have several biological, chemical, and physical properties that are distinct from natural forest soils. These properties include: modified soil organism activity, altered soil temperature regimes, disturbed nutrient cycling, higher soil pH, increased vertical and spatial variability, modified soil structure, anthropogenic materials (University of Manchester, 1996; Craul, 1985). The development of urban soils and the presence of physical disturbance caused by the incorporation of anthropogenic material affects the biogeochemical transformation of carbon (C) and nitrogen (N) by interfering with the interaction between key soil organisms and their processes (Lorenz and Lal, 2009; McDonnell *et al.,* 1997; Pickett *et al.,* 2001).

<u>Biogeochemical Cycling/ Urban Soil Processes</u>

Urban development and its associated effects on biogeochemical cycles, the ecology of landscapes, and regional and global climate, are growing increasingly important (Lorenz and Lal, 2009; Grimm *et al.,* 2000; Crutzen, 2004). In soils, the biogeochemical cycle refers to the cycling of nutrients, their biological, chemical, and physical properties, and the interactions between those properties (Curtis and Sloan, 2005; Turnbull, 2014; Totsche et al., 2009). Land that is converted for urban use is influenced by several direct and indirect factors that affect their soil physical, chemical, and biological properties. For example, the meso- microclimate, fauna, vegetation, and the built landscape vary significantly in each urban ecosystem (Konijnendijk *et al.*, 2006). Therefore, soil development is a continuing process, and because of the variety of the abiotic and biotic factors across the landscape, diverse soils are formed in different localities (Craul, 1985). These abiotic and biotic properties determine the quality of the soil for plant growth. However, once the soil and its properties are formed they are constant environments unless extensively disturbed by anthropogenic interaction or nature (Craul, 1985).

Direct Factors that Influence Soil Processes in Urban Areas

Direct factors that influence soil processes include incorporation of anthropogenic materials, covering of soil with impervious surfaces, and physical disturbances with machinery (Craul, 1999). Soil mixing prevents the proper distribution of crude material in soil, limits the depth and distribution of carbon and nitrogen, and soil structure and texture are altered (Craul, 1999). As a result, the pore volume and macropores are affected in the topsoil, subsoil layers become compacted but the degree of disturbance can be determined by the type of machinery used. Habitat communities for decomposers and microorganisms may be disrupted or destroyed (Lorenz and Lal, 2009; Byrne, 2007). Decomposer organisms that are native to topsoil may become buried while the subsoil organisms are unprotected from detrimental conditions like drought, temperature fluctuations, and solar radiation (Lorenz and Lal; Craul, 1999). There are advantages of soil mixing as the activity may increase the soil volume and total porosity thus improving habitat for soil microorganisms. Soil decomposer activity may be altered by shifts in soil pH which may affect C and N transformation (Lorenz and Lal, 2009; Beyer *et al.*, 1995). Urban soils tend to have higher pH values compared to natural forest soils. Elevated pH can be a result of released calcium (Ca) by weathered construction material, high concentrations of Ca and sodium (Na) on roadsides, or vegetation irrigation with enriched water. There are advantages and disadvantages associated with elevated pH values. Ideally, pH 7 will create a medium adequate for plant growth and enhance fertility; however, it may create issues for plants that prefer acidic environments. In addition, an overabundance of Ca or Na creates an imbalance with other elements and may prevent their uptake by roots (Craul, 1985).

Vegetation management practices in urban landscapes have a direct effect on biogeochemical cycling by the addition of chemical fertilizers to soil systems. Massive quantities of inorganic and organic fertilizers are frequently used in managed urban landscapes affecting N transformations (Baker *et al.*, 2001; Lorenz and Lal, 2009; Zhu *et al.*, 2004). Kaye *et al.*, (2004) explains that the conversion of native grasslands to urban use can have environmental consequences as fertilization and irrigation practices can lead to increased nitrous (N_2O) emission and decreased methane (CH_4) emission. In urban landscapes, C cycling rates are enhanced through increased irrigation and fertilization which increases soil respiration, aboveground net primary productivity, and total belowground C allocation (Lal and Lorenz, 2009; Pataki *et al.*, 2007). C and N cycling is also altered by the removal of organic material from the vegetated soil. For example, the in lawn care organic debris such as leaves and grass clippings are removed for a manicured look, therefore, reducing organic matter inputs into the soil (Craul, 1999; Lorenz and Lal, 2009). However, Byrne (2007) implies that the soil organic carbon pool may be higher because urban vegetation is often higher above- and belowground biomass compared to native vegetation. Soil erosion negatively impacts biogeochemical cycles by displacement and transport of soil particles along the soil surface (Lal, 2005; Lorenz and Lal, 2009). While soil displacement is natural process it is accelerated by urban development through surface soil removal, land conversion, construction, irrigation, etc. (Groffman *et al.*, 2003; Lorenz and Lal, 2009). Studies have shown that high rates of soil erosion and the consequent loss of C and N are a result of vehicular traffic and excavations (Crim *et al.,* 2011). In addition, altered water quality caused by erosional losses of phosphorus (P) can happen during urban development (Lorenz and Lal, 2009).

Anthropogenic activities also affect soil properties and chemical cycling through soil compaction. Gill in 1971 stated, "*Soil compaction is a worldwide problem of enormous economic and ecological importance. In the U.S. alone it was estimated that in 1971 soil compaction accounted for an annual loss in crop values of $1.2 billion.*" As the global population increases, there is a growing concern that even more, severe levels of anthropogenic- induced soil compaction will occur (Kozlowski and Pallardy, 1997). Increases in bulk density because of compaction, is not limited to the topsoil but penetrates to a considerable depth and decrease soil porosity in the subsoil (Lehmann and Stahr, 2007; Lorenz and Lal, 2009). As a result, there is a restriction of water and air permeability and storage capacity that may C and N cycling processes (Lal, 2005; Lorenz and Lal, 2009). The translocation of C and N may be altered by soil compaction because roots and soil biota cannot penetrate to subsoil horizons where important processes take place. (Harris, 1991). During the construction of impervious surfaces, soils are often de-surfaced, filled, and compacted thereby altering the biogeochemical cycling (Mullins, 1991; Kaye et al., 2006).

Indirect Factors that Influence Soil Processes in Urban Areas

Indirect factors of urbanization include UHI effects, atmospheric deposition of pollutants, soil hydrophobicity, modification of vegetation structure and composition, and the introduction of nonnative plant species (Craul 1992; Pickett et al., 2001). Faulker (2004) states that urbanization leads to changes in forest structure and composition in both the canopy and understory. Anthropogenic activities modify vegetation structure and composition by removing native vegetation and replacing it with non-native plant species that would not occur in natural landscapes (Byrne, 2007). Misguided vegetation management practices have led to the introduction of non-native, invasive, exotic plants in urban areas that reduces the productivity of native urban vegetation (Hope *et al.,* 2003). The quality of quantity of plant litter which is the main source of soil organic matter is critical to biogeochemical processes (Kaye *et al.,* 2006). For example, the lack of plant litter on mowed or artificial lawns are contribute negatively to greenhouse gases than natural vegetation covers that they replace (Koerner and Klopatek, 2002). Pressures of urban activities on soil decomposer communities can drastically alter the condition of organic matter. Typically, in urban soil environments, C and N relationship dynamics are complicated by soil disturbances, therefore, affecting chemical processes essential organisms and their processes (Lorenz and Lal, 2009).

Soil Pollution in Urban Environments

Historically, soil surveying and research primarily focused on geochemical processes in agricultural and forest soils rather than urban and suburban soils (De Kimpe and Morel, 2000). Recently, the need for assessing soil properties in urban and suburban soils has expanded because of the growing public concern about the environment and human health. One of the challenges of global change and growing populations is the influence of soil on human health. There are several examples in the literature of how soil quality has negatively impacted human health. Polizzotto et al., (2008) documents the release of arsenic to groundwater by redox cycling in the soils of South East Asia. Patz et al., (1998) studies the impact of soil moisture on the spread of malaria. Seneviratne et al., (2006) examines the severity of fatal heat-waves in Europe due to the reduction of the soil moisture buffer. Several authors have identified urban soils as a major source of lead (Pb) exposure in children (Sheppard and Evenden, 1994; Lanphear and Roughmann, 1997 Thornton *et al.,* 1994). A study in the United Kingdom revealed that ingestion because of hand-to-mouth activity accounted for 50% of a child's Pb intake (Thornton, 1990). Particularly, children are exposed to elevated pollution levels in public parks and playgrounds where dust from the ground has toxic effects because of inhalation or ingestion. Children are especially susceptible to Pb poising to due to their developing nervous system and high absorption rate (Figueiredo, et al., 2009; Manta et al., 2002).

Urban soils receive substantial amounts of metals from a variety of sources such as traffic, industrial, and domestic emissions (Wei and Yang, 2010). These metals are referred to as trace metals because they are present at a background level. Lead, cadmium, zinc, mercury, Although these metals can be necessary or beneficial to plants at certain levels, they can also be harmful when exceeding specific thresholds (Wang, 2005; Facchinelli *et al.*, 2001). These metals are unique because unlike many organics and radionuclides, they do not degrade with time and are persistent in the terrestrial environment. Trace metal balance influence soil quality and ultimately the ecological and agricultural functions of soil. Furthermore, trace metal balance soils critically important to soil contamination and pollution as these metals persist much longer in the soil than any other compartments of the biosphere (Kabata-Pendias,1995). Abiotic and biotic factors such as pH, cation exchange capacity, and soil organic matter can affect the availability of trace metal availability in the soil (Jean-Philippe *et al.*, 2011). Degradation of soil pollutants depends on several factors, the chemical structure of the pollutants, composition and catabolic activity, and the native microbial community (Turnbull, 2014; Reid *et al.*, 2000). Micro-organisms also use pollutants as nutrients because soil microbes can break down many harmful pollutants and utilize them for their benefit (Turnball 2014, Beyer *et al.*, 1995). Chemical pollutants are abundant in urban environments thus microbes that degrade pollutants may thrive in those environments (Turnbull, 2014; Beyer *et al.*, 1995).

Brownfields and Greyfields

The Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) defines a brownfield site as "*real property, the expansion, redevelopment or reuse of which may be complicated by the presence of a hazardous substance, pollutant, or contaminant*" (www.osha.gov). Brownfields are a local environmental consequence to a changing industrial landscape influenced by technology change, economic transition, and global competition. As described by Bjelland (2000), capital follows innovation, however, toxic residue in soil and groundwater may persist long after industrial processes have ceased. With more than 450,000 brownfields in the U.S. that are currently undergoing or require remediation, it is imperative for cities to restore and recycle brownfield sites to ensure they're economic, environmental, and social health (Bjelland, 2000; Overview of EPA’s Brownfield Program, 2018). Another environmental consequence of economic transition in urban landscapes is Greyfields. Greyfields are previously developed properties typically buildings with existing infrastructure and utilities—although the previous use is obsolete and the structures are in disrepair. Legacy impacts from Greyfield presence can cause ecosystem disruptions such as flooding, erosion, stream or wetland disturbance, habitat destruction or fragmentation (Merritt, 2006). However, Greyfields present opportunities to revitalize the built environment and reconnect the natural environment through daylight streams, restore wetlands, repair and reconnect habitats, thus making them more sustainable and economically productive (Newton et al., 2012).

Statement of Problem

From 1942 through 1948, the federal government acquired approximately 55,000 acres (22,257 hectares) of land in Oak Ridge, TN to accommodate facilities that would produce the materials as a part of the World War II Manhattan Project. Land that was formerly rural and agriculture was quickly converted to support the facilities, individuals and their families moving to the area for work. In 1943, construction began on the X-10 nuclear research facility, which housed the world's first nuclear reactor. The X-10 site would later become known as the Oak Ridge National Laboratory (ORNL). Today, ORNL is the largest and most diverse research and development institution in the Department of Energy (DOE) system. ORNL is a unique facility, often referred to as a "small city" because it has its own energy supply, facility services, regulatory groups, and governing body that determine its operations, use, and allocation of resources. The land surrounding the ORNL campus is intensively used for multiple purposes (i.e. research facilities, utility infrastructure) to meet the national goals and objectives of DOE. Because of those activities, the land has experienced severe physical disturbance along with inputs of excess heavy metals and toxic chemical wastes associated with its historical and contemporary use. Like any urban environment, ORNL has numerous challenges that interfere with its progress toward environmental sustainability. Some of the issues affecting ORNL resemble those affecting urban communities worldwide (i.e. land-use change and development, land erosion, soil contamination and compaction, altered vegetation, forest pest, and invasive plants, etc.). Furthermore, this "small city" of ORNL is a representative of a community where environmental pollution problems could hinder its quest toward environmental sustainability.

Over the past seven decades, the activities at ORNL have shifted and policymakers have worked to make ORNL environmentally sustainable through effective and strategic planning. To guide the future of environmental resources and sustainable landscape practices on ORNL's campus stakeholders have developed the "Sustainable Landscape Initiative Plan 2020". One of the objectives of the Sustainable Landscape Initiative 2020 was to inventory and assess the vegetation present on the ORNL campus and quantify the environmental services associated with the vegetation (Gardner *et al.,* 2011) An urban tree inventory and assessment will provide an overview of urban forest structure, species condition, and distribution that are necessary to quantify the environmental benefits and economic values associated. Second, a soil study is necessary to gather baseline data about the current conditions chemical of the soils that support the landscape vegetation across the campus. Knowledge of this information can help better inform policy and management decisions for vegetation present on campus.

Objectives and Hypotheses

The objectives of this research study were to quantify the structure, function, and economic value of the landscape trees present on the ORNL campus and characterize soil properties. **Research Objective 1:** Assess vegetation in managed landscapes to quantify the environmental benefits and landscape value. To quantify forest structure across ORNL campus, a field study that consist of a complete tree inventory was conducted to investigate (1) tree species diversity, (2) diameter at breast height (DBH) distribution and (3) tree condition. The data from this inventory were used in the i-Tree Eco application to estimate the environmental benefits and monetary value of landscape trees. **Hypothesis 1:** Trees in highly disturbed areas will have lower condition class ratings than trees in less disturbed areas. **Hypothesis 2:** Environmental effects and Importance Values will be strongly correlated with tree size (DBH, leaf area, canopy size) rather than the relative abundance (number of trees). **Research Objective 2:** Characterize the chemical soil composition within managed vegetation sites on the ORNL campus. A soil assessment was conducted to assess 10% of landscape vegetation below-ground environment. Soil elemental concentrations were determined using a total dissolution method along with inductively coupled plasma optical emission spectrometry.

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PART II

i-TREE ECO ANALYSIS OF LANDSCAPE VEGETATION ON REMEDIATED AREAS OF OAK RIDGE NATIONAL LABORATORY

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Abstract

The Oak Ridge National Laboratory (ORNL) is the largest and most diverse energy, research, and development institution within the Department of Energy (DOE) system. As such the site endures constant land development that creates rigorous growing conditions for urban vegetation. Natural resource managers at ORNL recognize that trees are an integral component of the landscape and are interested in characterizing the urban forest and their associated ecosystem services benefits. To determine the structure, function, and economic value of the urban forest, an urban tree assessment was conducted during the summer and fall of 2017. We employed i-Tree Eco methodology for data collection that included geolocation, tree species, diameter at breast height (DBH), and tree condition ratings. Further, native and non-native trees with a DBH >7.62 cm were included in this assessment. Our study reveals 1160 trees, composed of 62 species, and 30 genera. The most abundant species were *Acer rubrum* (10.7%), *Cercis canadensis* (9.6%), and *Quercus palustris* (6.3%). The most important species in terms of population size, leaf area size, and environmental effects were *Acer rubrum* (19.3), *Quercus palustris* (17.4), *Juniperus virginiana* (15.0), *Pinus strobus* (11.2), and *Quercus phellos* (7.1). The structural value of urban trees equates to 2.02 million dollars. Assigning a monetary value on urban forest benefits help to inform decisions about urban forest management, ideally on cost-benefit analysis.

Keywords: Urban tree(s), urban forest, urban vegetation, landscape vegetation, tree inventory, tree assessment, ecosystem service(s), ecological function, i-Tree Eco, economic valuation

Introduction

Proper management and stewardship of an urban forest require an inventory of the resource (Miller *et al.*, 2015). An inventory is an initial part of the short and long-term assessment and monitoring of urban tree populations (Nowak *et al.*, 2008). Urban forest inventories provide information on the structure, condition, and management needs of the resources. There are challenges to account for all benefits and cost for proper management of an urban forest. Some vegetative cost such as planting, maintenance, and removal, are easy to track by municipalities. However, estimating tree benefits such as ecological functions can be more difficult to determine without a substantive inventory and valuation system. A common goal of urban tree inventories and assessments is to use the information to quantify the ecosystem services. Programs that measure ecosystem services are valuable tools for urban forest managers, researchers, and municipal forestry programs.

Generally, there are two types of urban forest assessments, top-down and bottom-up assessments. A top-down assessment uses aerial photography, geographic information system (GIS), or remote sensing-based tools for analysis to describe urban tree canopies (Nowak, 2013). Aerial imagery is accessible through open sources like Google Earth and Google Maps. Aerial imagery allows users to gather a more complete description of the urban forest by observing the canopy in different areas and dates (spatial analysis). GIS programs are more complex because they often require licensing, however, programs such as ESRI ArcMap are useful for importing aerial imagery, that can be used to create and analyze maps, manage and store geographic information. Also, GIS applications can be useful for calculating vegetative indices to determine whether the target being observed consist of green vegetation or not. Remote sensing is a type of geospatial technology used to map and monitor urban canopy cover and other landscape features.

This tool can track ecosystem features such as carbon storage, urban heat island (UHI) effect, and tree canopy mortality. The accuracy of the top-down assessment depends on factors like image quality, spatial resolution, and technology performance (American Forest: Urban Forest Assessment Resource Guide, 2013).

Specific software tools that can be used in a top-down assessment include: high-resolution imagery of an urban tree canopy that uses digital imagery gathered from satellites i.e. NASA Landsat program or National Agriculture Imagery Program (NAIP). Integrated Valuation of Environmental Services and Tradeoffs (InVEST) is a host of open-source tools that allow users to input GIS data such as land use, land cover, and topography and it produces maps that model results in their biophysical or economic terms. Light Detection and Ranging (LiDAR) is a valuable resource for characterizing and monitoring trees in an urban environment. LiDAR relies on reflected light and sensors emit their own energy in the form of a laser. Incorporating LiDAR into tree canopy assessments can improve the ability to detect smaller or recently planted trees, resulting in a more accurate representation of a city's tree canopy (American Forest: Urban Forest Assessment Resource Guide, 2013).

Bottom-up approaches use tree inventory and field data to estimate ecosystem services produced by an urban forest (Nowak, 2013). These assessment tools are beneficial because they provide ecosystem value, benefits information, and help to guide management decisions for an urban forest (American Forest: Urban Forest Assessment Resource Guide, 2013). There are two sampling methods to conduct a bottom-up assessment which are complete and sample inventory. Complete inventories use require data from every tree in an assessment area. Complete inventories also are used for daily management of street trees or trees that occur in public areas. Complete inventories are useful for observing pest and disease occurrences. A sample inventory generally looks at 3-6 % of the full assessment area and are scaled-up to estimate the structure of a forest. Sample inventories are used when the assessment is too large or there aren't adequate resources to conduct a complete inventory. Specific software tools that use a bottom-up assessment include i-Tree Eco and i-Tree Streets model (www.iTreetools.org). These assessment tools can quantify forest structural attributes (number of trees, species composition, health, and density), stormwater mitigation information, carbon storage capabilities, hourly air pollution removal, energy savings, and property value increase (Nowak et al., 2008). There are models for individual tree benefit estimates such as the Center for Urban Research Tree Carbon Calculator (CTCC) developed by the U.S Forest Service (American Forest: Urban Forest Assessment Resource Guide, 2013). The CTCC is a tool that provides information on carbon benefits of individual trees based on characteristics of the region or climate zone in which the tree is located. The program uses information from 16 climate zones in the U.S. along with tree attributes to estimate biomass capacity, annual sequestration rates, and lifetime benefits (American Forest: Urban Forest Assessment Resource Guide, 2013). Lastly, the National Tree Benefit Calculator is an accessible tool that estimates environmental benefits and monetary value of an individual tree with inputs of species and size along with the location (zip code) (American Forest: Urban Forest Assessment Resource Guide, 2013).

The Oak Ridge National Laboratory (ORNL) is an active federal research facility that includes contaminated areas and structures used during the Manhattan Project (1942-1948) to present. Originally used for the production and separation of plutonium during World War II, significant amounts of chemical waste were deposited into the soil, buried, and directly discharged into local waterways (Jean-Philippe, 2010). The U.S. Department of Energy (DOE) and Environmental Protection Agency (EPA) are dedicated to the cleanup of the Bethel Valley facilities, which include remediating contaminated soils, sediments, water, and infrastructure (UCOR, DOE OREMP, 2018).

Across the ORNL campus, the destruction of many of the facilities has led to the presence of brownfield and greyfields. ORNL seeks to enhance the campus environmental experience and increase their overall campus sustainability, by developing and maximizing the benefits the urban forest provides. We investigate the overall condition of trees planted in areas where legacy contaminates material are present. Specific research objectives were to assess vegetation in managed landscapes to quantify the environmental benefits and estimate tree value. To do this, forest structure was quantified across ORNL campus, by conducting a field study and collecting tree species diversity, diameter at breast height (DBH) distribution and tree condition (good, fair, poor, dead). We hypothesized trees in highly disturbed areas will have lower condition class ratings than trees in less disturbed areas. Environmental effects and Importance Values (IV) are hypothesized to be strongly correlated with tree size or canopy size rather than the number of trees present within that given species.

Methods

Site Description

The Oak Ridge National Laboratory is located on the Oak Ridge Reservation (ORR) in Anderson and Roane County Tennessee, USA. The ORR is located in the temperate region and has four distinct seasons. The average low temperature is 8.4 °C, the average high temperature is 20.8°C, and annual precipitation is 139.9 cm (Data United States Climate, 2018). According to the Roane County Soil Survey (USDA, 1942 and 2002), the general soil types found in the Bethel Valley was generally classified as either Colbert Series or Upshur Series silty clay loams. The dominant forest cover type on the ORR is oak-hickory, mixed forest, and conifer (Parr *et al.,* 2015).

The ORNL site occupies approximately 1808.9 ha (4470 acres) and includes facilities in two valleys- Bethel Valley and Melton Valley (Figure 2.1). Bethel Valley is the site of the main campus area which has over 190 buildings and over 4,500 daily occupants. The site has many different land uses including research laboratories, brownfields, greyfields, undeveloped areas, and natural areas. The ORNL main campus where this tree inventory was conducted constitutes approximately 190.2 ha (469 acres).

Inventory Methods

The tree inventory method was developed collaboratively by the ORNL Natural Resources Management Program and the Department of Forestry, Wildlife and Fisheries at the University of Tennessee. Inventory methods were adopted from the i-Tree Eco protocol (i-Tree Eco Field Guide Manual v. 6.0). Attributes collected include tree ID, species, diameter at breast height (DBH) (>7.62cm), tree conditions (good, fair, poor, critical, dead), and geographic coordinates. The landscaped areas of the campus were divided into nine sections and assigned a unique Block ID (B1, B2, B3, B4, B5, B6, B7, B8, and B9) (Figure 2.2).

Data Collection

All data were collected following the i-Tree Eco complete inventory protocol (i-Tree Eco Field Guide Manual v. 6.0). A Trimble GeoExplorer 6000 series was used to collect and store inventory data. Data collected was downloaded from the GPS unit to a desktop computer daily using the Trimble GPS Pathfinder Office version 5.81, 2015. MapInfo 15.0 created by MapInfo Corporation was used to create maps. Once all data collection was completed, it was sent to the USDA Forest Service Northern Research Station for i-Tree Eco analysis.

Figure 2.1. The ORNL main campus study site in Oak Ridge, TN.

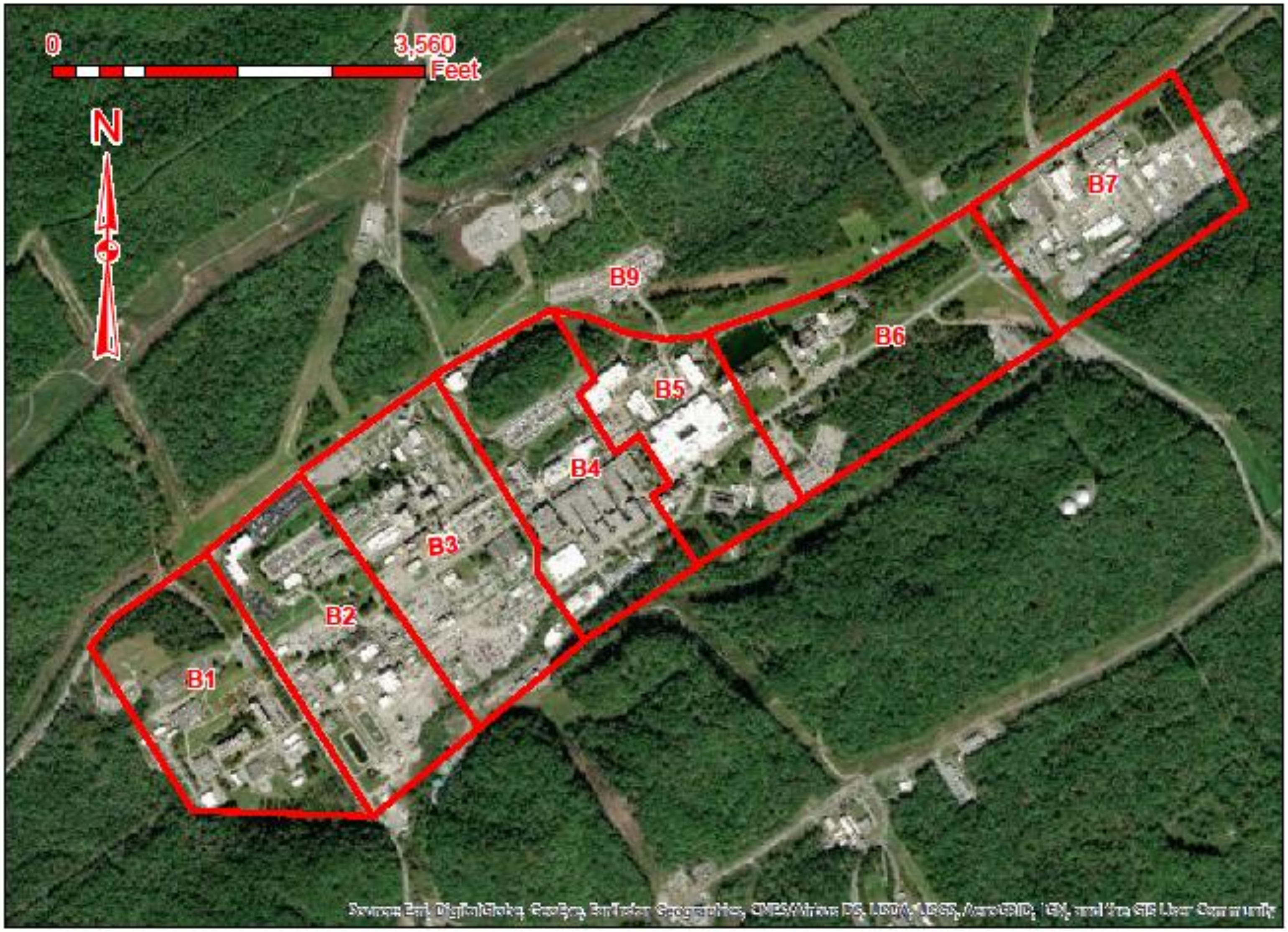

Figure 2.2. The study site outlined in red for the nine blocks inventoried.

i-Tree Eco Analysis

i-Tree is a suite of forest analysis and benefit assessment tools developed by the United States Depart of Agriculture Forest Service (www.itreetools.org). The i-Tree suite consists of multiple analysis and benefit assessment tools that provide information on urban and community forest that can aid in forest management and advocacy. The specific software used in this study is i-Tree Eco, previously called Urban Forests Effects (UFORE) model (Nowak & Crane 2000). I-Tree Eco uses standardized field data from a sample inventory or complete tree inventory along with inputs of local air pollution rates and meteorological data, to quantify urban forest structure, environmental benefits and services, and the monetary value of these services (www.itreetools.org; McPherson and Simpson, 2002; Maco and McPherson, 2003). An assessment of ORNL urban trees was conducted to quantify the same attributes. Data from 1160 trees located throughout the ORNL campus were analyzed using i-Tree Eco model version 6. The i-Tree Eco model utilized reported weather and pollution estimates from 2013 a local station in Anderson Country, TN. Importance Values (IV) as provided by i-Tree Eco model are calculated as the sum of the percent population and percent leaf area of a given species.

Statistical Analysis

Statistical analysis for this assessment was performed using SPSS 25. A Pearson correlation coefficient was computed to determine which tree attributes were more influential on environmental effects. Tree attributes such as diameter, leaf area, and canopy size were used as independent variables to predict water interception and avoided stormwater runoff. A one-way Analysis of Variance (ANOVA) was used to determine differences in condition ratings according to the block location of the tree site.

Results and Discussion

Urban Forest Structure

To help characterize the urban forest structure we conducted a bottom-up assessment and analyzed the data to provide details about species diversity, diameter distribution, and tree conditions. ORNL managed urban tree population is comprised of 1160 trees, with a total of 62 species, and 30 genera (Figure 2.3). The most abundant species are *Acer rubrum* (Red maple, 10.7%), *Cercis canadensis* (Eastern redbud, 9.6%), *Quercus palustris* (Pin oak, 6.3 %), *Acer saccharum* (Sugar maple, 6.2%), and *Juniperus virginiana* (Eastern red cedar, 5.8%) (Figure 2.4). Additionally, trees were categorized into diameter classes to illustrate the proportion of trees at various stages of maturity. Small trees with diameter <15 cm constitute 34.4% of the population, whereas mid-size trees with DBH 15 and 46 cm account for 47.7%, and 17.9% of the population are large/mature trees with a DBH larger than 46cm (Figure 2.5). The mean DBH is 10.97cm and the maximum DBH is 114.8 cm for a *Quercus phellos* (Pin oak) tree. Condition class ratings were given a numerical value good-4, fair-3, poor-2, dead-1. Overall tree condition ratings were categorized as 79.4% good, 15.6% % fair, 2.4% poor, 1.9% critical, and 0.7% dead (Figure 2.6).

Tree condition rating

Condition rating: An assessment of the tree's structural integrity and health at the time of appraisal. Good (4) demonstrates no apparent problem with a tree's structure, fair (3) demonstrates minor problems with tree's structure, poor (2) major problems with a tree, critical/dead (1) extreme problems. Factors that impact structural integrity include broken or dead branches, decay, codominant stems, included bark, broken or dead roots, asymmetrical growth, and potential for failure in the future. Other issues factor that were considered in condition rating includes

Figure 2.3. The distribution of urban trees (green dots) located across the ORNL campus.

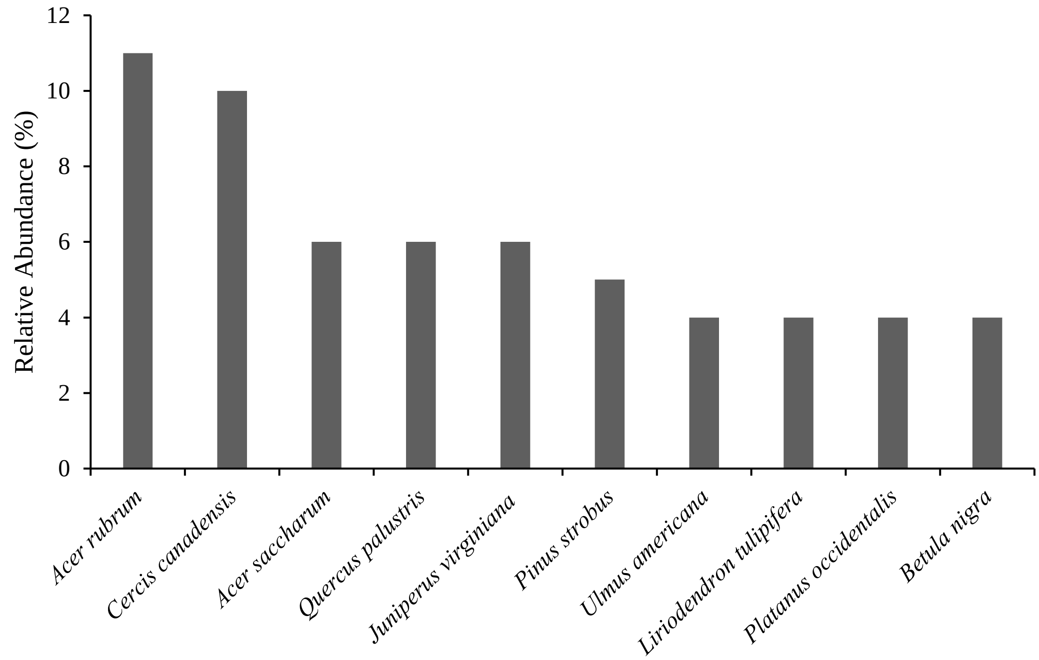

Figure 2.4. The most dominant species and their relative abundance of Trees across ORNL campus.

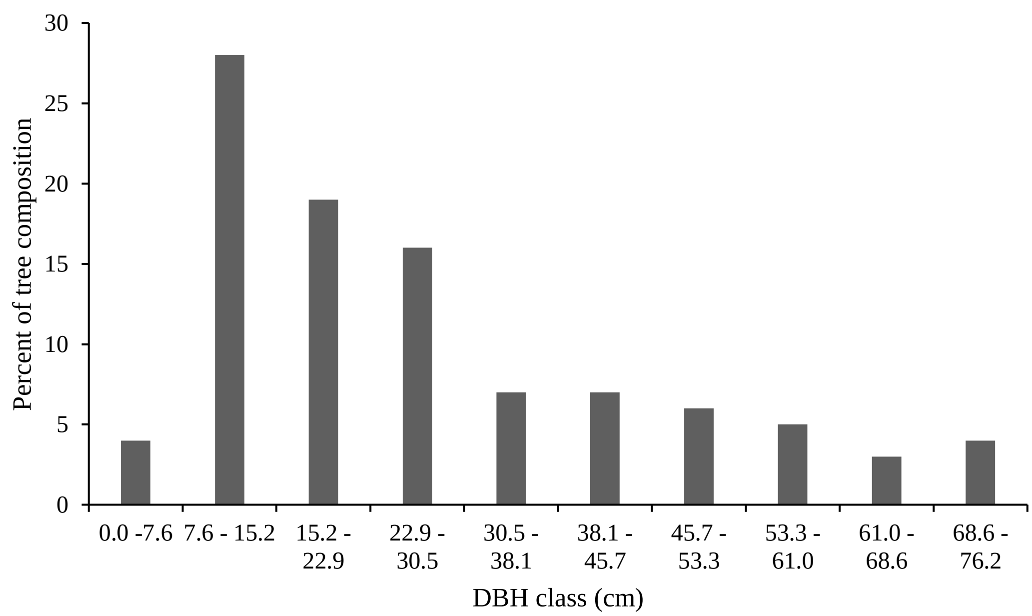

Figure 2.5. Percent tree population by diameter class.

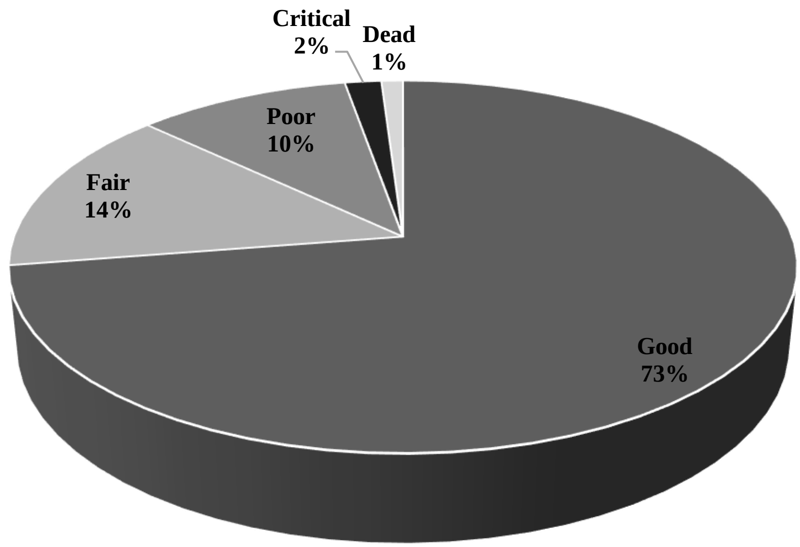

Figure 2.6. Overall tree condition ratings (good, fair, poor, critical, dead) determined for all inventoried trees.

abiotic disorders, physical injuries, chemical damage, limited growing space, improper installation, and poor maintenance practices. (Ingram, 2000; Cullen, 2005). To determine which Block (B1, B2, B3, B4, B5, B6, B7, B8, and B9), had higher or lower tree condition ratings, a one-way ANOVA was used to compare means according to tree block location. We hypothesized that trees in highly disturbed areas will have lower condition ratings. The trees with the best condition ratings were in B4, B5, and B6 (Figure 2.7). The mean condition for B4, B5, and B6 was significantly different from all other blocks when tested at the (α=0.05). For example, B4 differs from B1-2, while B5 differs from B1-B3, and B9; and B6 differs from B1-B3.

Leaf Area and Importance Values

Most of the environmental benefits derived from trees are attributed to their leaf surface area, which is contingent with size and vegetative growth characteristics (Livesley *et al.,* 2016).

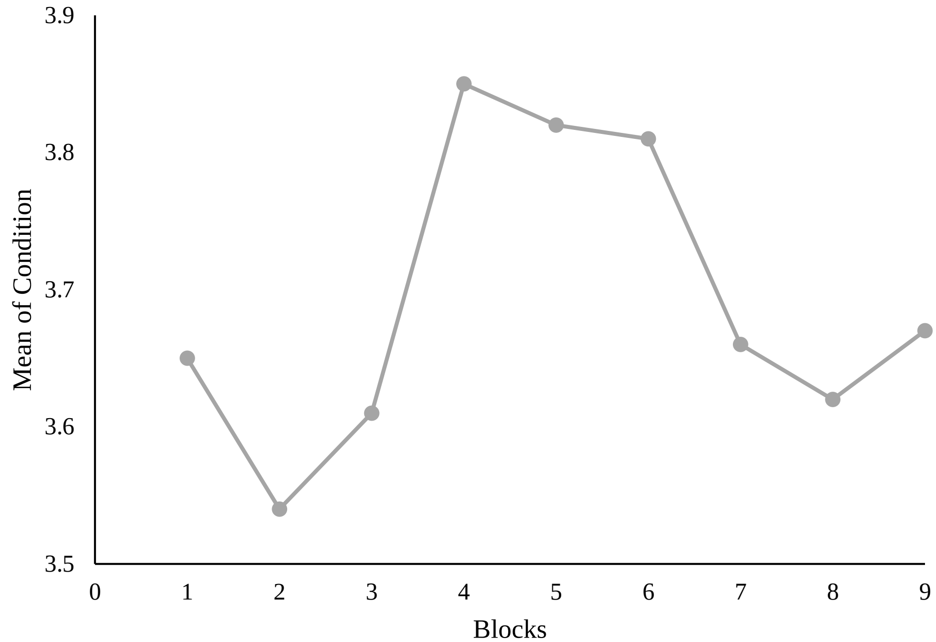

Figure 2.7. A comparison of condition ratings using the mean of conditions for Block 1-9.

Leaf area is a function of stem diameter size because as diameter increase so does the crown size. Leaf area (m^2) is estimated by the i-Tree Eco model and for this study area, the canopy cover is estimated to be 4.9 ha of land area and 23.36 ha of leaf area. The leading species with the largest proportion of total leaf area are pin oak (11.1%), eastern red cedar (9.2%), and American sycamore (8.7%). Leaf area amount for a species is a determinant of its significance or importance in an inventory. An Importance Value (IV) is calculated as the sum of the percent population and percent leaf area of a given species. Red maple (19.3), pin oak (17.4), eastern redcedar (15.0), sugar maple (13.9), eastern redbud (13.4) are the leading species with the greatest IV (Table 2.1).

Red maple exceeds all other species in IV because aside from being the most abundant species within the inventory, it has the fourth-highest percent leaf area (Figure 2.8). However, in terms of DBH distribution, a third of the species (35%) are small/ young trees with an average DBH of 19.9 cm. Thus, this species has yet to reach optimum benefit output (Figure 2.9). Pin oak is the third most abundant species, has the second-highest IV, and accounts for the highest proportional amount of leaf area compared to any species: 279,862 square feet of leaf area or 11.1 % of the total leaf area. Eastern redcedar is the fifth most populous species and constitutes 226,402 square feet or 9.2% of the leaf area and has the third-highest IV. For comparison, eastern redbud is the second most abundant species, more numerous than pin oak and eastern redcedar but accounts for a total of 96,875 square feet or 3.8% of leaf area (Table 2.1). A high relative abundance does not suggest that this species is providing more environmental effects nor does a high importance value suggest that a particular species should be encouraged in future plantings; rather these species are currently the most abundant in the urban forest structure.

<u>Structural and Functional Values</u>

An advantage of a valuation model such as i-Tree Eco is the capability to provide an estimate for structural and functional value of vegetation. Structural value is the monetary value of a tree based on its physical attributes and the replacement cost for a similarly sized tree if it were removed from the landscape. Individual tree structural values are calculated using the Council of Tree and Landscape Appraisers (CTLA) formula (2000). The CTLA formula incorporates tree

Table 2.1. Ranking of species by Importance Values. urban forest i-Tree Eco Assessment.

Species Name	Percent Population	Percent Leaf Area	IV
Acer rubrum	10.7	8.6	19.3
Quercus palustris	6.3	11.1	17.4
Juniperus virginiana	5.8	9.2	15
Acer saccharum	6.4	7.6	13.9
Cercis canadensis	9.6	3.8	13.4
Platanus occidentalis	3.9	8.7	12.7
Pinus strobus	5.4	5.8	11.2
Liriodendron tulipifera	4.2	4.1	8.3
Ulmus americana	4.2	3.9	8.1
Quercus phellos	2.6	4.5	7.1

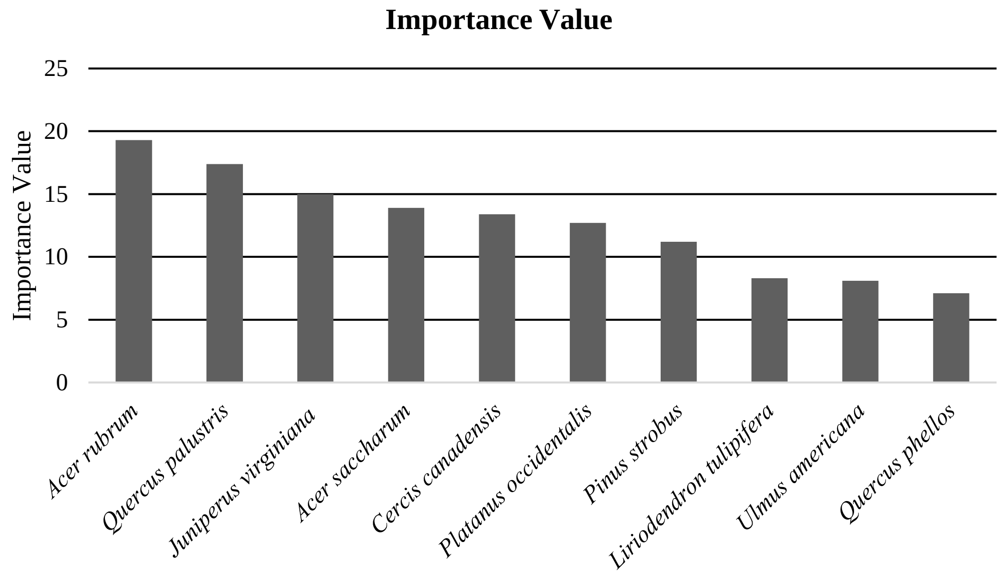

Figure 2.8. Top ten tree species in descending order of Importance Values.

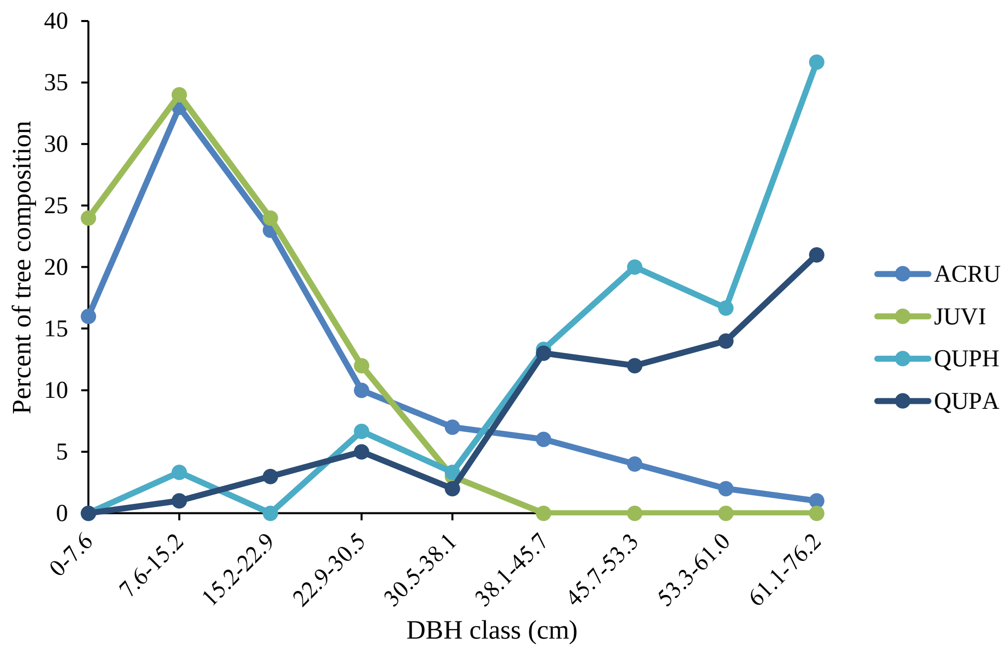

Figure 2.9. Distribution of diameter size in four top performing species: *Acer rubrum, Juniperus Viriginiana, Quercus phellos, Quercus palustris* population

species, diameter, condition, and location information (*Value = Basic Tree Cost x Species Rating % x* Condition Rating % x Location Rating %) (Nowak et al., 2002; Gooding et al., 2000). For the trees counted in this inventory, the structural value of all species is estimated to be \$2.02 million with willow oak, eastern red cedar, pin oak, red maple, and eastern white pine as the leading species with the greatest structural value (Figure 2.10). The average structural value for per willow oak is \$7,511, followed by eastern red cedar at \$3,054 (Table 2.2). The structural value and annual functional values are related to the quantity, size, and health of trees as illustrated in Table 2.2.

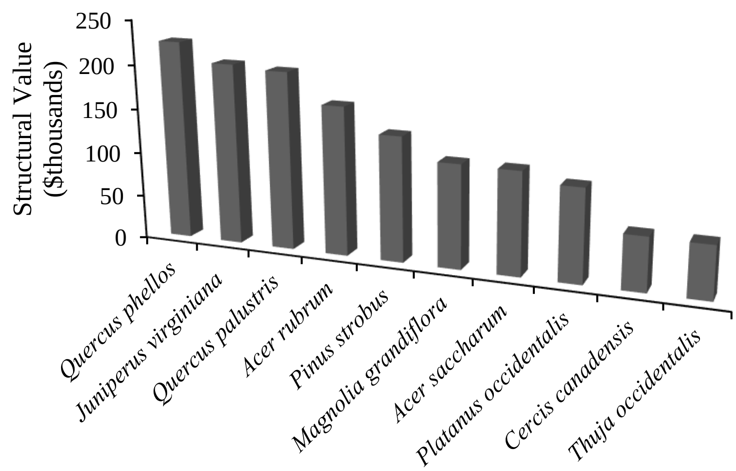

Figure 2.10. Top ten species with the greatest structural value.

Table 2.2. Average values for characteristics of the top ten most abundant species.

Species	**% of All trees**	**Avg. DBH (cm)**	**Average Structural Value ($)**
Acer rubrum	10.7	19.99	1,357
Cercis canadensis	9.6	14.24	549
Acer saccharum	6.4	27.35	1,582
Quercus palustris	6.3	53.09	2,789
Juniperus virginiana	5.8	38.36	3,054
Pinus strobus	5.4	35.57	2,265
Liriodendron tulipifera	4.2	17.97	964
Ulmus americana	4.2	22.23	967
Platanus occidentalis	3.9	39.81	2,332
Quercus phellos	2.6	60.09	7,511

Carbon Storage and Sequestration

Carbon dioxide (CO_2) is among the more prominent pollutants of concern in the global climate crisis. Increased atmospheric CO_2 is attributable mostly to fossil fuel and industrial processes, along with agriculture and deforestation operations worldwide (U.S. EPA Global Greenhouse Gas Emissions Data, 2017). In 2017, the global average atmospheric CO_2 was 405.0 parts per million (NOAA, 2019). Naturally, forests serve as a carbon sink by absorbing carbon during photosynthesis, storing carbon as biomass in above-and below ground structures, and producing oxygen as a by-product of photosynthesis (Bellassen & Luyssaert, 2014). Increasing the number of trees has the potential to mitigate the accumulation of atmospheric carbon (Myneni et al., 2001). Urban areas particularly contribute to atmospheric pollution from mobile and stationary sources and continued development. Thus, urban forests perform a vital ecosystem service by sequestering and storing CO_2 (Hoornweg, 2012). i-Tree Eco estimates carbon storage and gross carbon sequestration values are calculated based on the price of $143 per ton, a value determined by i-Tree Eco. The estimated gross sequestration of all trees in this assessment was estimated to be 9.5 tons of carbon per year with an associated value of $1,360. In addition, they are estimated to store 320.6 tons of carbon amounting to $45,800 in annual benefits. Of the species sampled, pin oaks and willow oaks accounted for approximately 35.6% of the total carbon stored and 27.1% of all sequestered carbon due to of their relative abundance, large structure, and biomass capacity.

Air Pollution Removal

Air pollution effects were estimated using species characteristics, the amount of leaf biomass, and recent pollution and weather data available (base year 2013). The i-Tree Eco model estimated that trees mitigate 278 kg of air pollution which include Ozone (O_3), carbon monoxide (CO), nitrogen (NO_2), and particulate matter less than 2.5 microns ($PM_{2.5}$), and sulfide oxide (SO_2)

per year with a minimum value of $65.7. Pollution removal value is calculated based on the prices of $1.6 per 2.2 lbs. (CO), $0.126 per 2.2 lbs. ($O_3$), $0.014 per 2.2 lbs. ($NO_2$), $0.005 per 2.2 lbs. ($SO_2$), $3.7 per 2.2 lbs. ($PM_{2.5}$). While there were positive effects in relation to air pollution abatement there were also negative effects. On an annual basis, trees are estimated to emit 568.57 lbs. of volatile organic compounds (VOCs) that includes 430.6 lbs. of isoprene and 138.1 lbs. of monoterpenes. Two of the most important species (pin oak and willow oak) generated 54% of VOC emissions which have negative effects in relation to ozone formation. Increased ozone has negative effects in terms of human respiratory health.

Stormwater Benefits

Stormwater management is an area of concern among environmental managers and engineers, due to the potential of flooding following heavy rainfall. Further, it can contribute to pollution in streams, lakes, and rivers where they harm water quality. Stormwater runoff occurs when there is a significant amount of precipitation that isn’t captured by the tree canopy and isn’t absorbed by soil thus it becomes surface runoff (Hirabayashi, 2012). Urban trees and shrubs are essential in reducing runoff volumes because they capture precipitation in their canopies, while their root systems are able to filtrate and store water in the soil. All components of the tree physical structure such as leaves, branches, and bark are essential in capturing precipitation, however, only the amount retained by leaves was accounted for in this analysis. Precipitation interception was estimated using rainfall totals in the base year 2013.

The total annual precipitation (172.05cm) in 2013 was slightly higher than the reported precipitation totals (156.2 cm) in 2017 during the year of this study (www.ncdc.noaa.gov/cag/). Trees recorded in this inventory were estimated to intercept 4,711 m^3 (1,244,620 gallons) and

Table 2.3. Avoided runoff values for species with the greatest overall impact on stormwater mitigation.

Species Name	Number of Trees	Leaf Area (ha)	Water Intercepted (m^3/yr)	Avoided Runoff (m^3/yr)
Quercus palustris	72	2.59	522.43	114.81
Juniperus virginiana	67	2.14	432.6	95.07
Platanus occidentalis	45	2.04	411.69	90.47
Acer rubrum	123	2.02	406.57	89.35
Acer saccharum	73	1.77	356.41	78.32
Pinus strobus	62	1.36	274.89	60.41
Quercus phellos	30	1.77	213.2	46.85
Liriodendron tulipifera	48	1.36	195.09	42.87
Ulmus americana	48	1.06	183.91	40.42
Cercis canadensis	110	0.97	180.23	39.61

helped to mitigate runoff by an estimated 1,035m3 (273,418 gallons) per year with an associated value of $2,440. In Table 2.3 Pin oak, Eastern redcedar, and American sycamore are among the top-performing species for stormwater benefits. The avoided runoff value is calculated by the price of $2.361/m³, a value assigned by i-Tree Eco. The i-Tree Eco model estimate results reveal that size (DBH, leaf area, and canopy size) are most important in terms of rainfall interception (Table 2.3).

To further investigate the association between DBH, leaf area, species abundance, and environmental effects, a stepwise regression was used to determine whether leaf area (m^2) and DBH were correlated with water interception and avoided stormwater runoff. There was a strong correlation between Diameter at Breast Height and water interception, $r = .88$, $p = \leq .001$. However, species abundance was moderately negatively correlated with rainfall interception ($r = -.57$) (Table 2.4). Results were similar for avoided runoff (Table 2.4). The number of trees didn't change nor improve the model prediction of anticipated rainfall interception. This suggests that the DBH which is a predictor of leaf area and canopy size is the most important tree attribute that influences environmental effects particularly, hydrology effects (Table 2.3). Hence, it explains why species

Table 2.4. Pearson correlation coefficient for the relation of avoided runoff, water interception, species count, leaf area, and DBH.

	Avoided runoff (m^3/yr)	Species count	Leaf area (m^2)	DBH (cm)
Avoided runoff (m^3/yr)				
Species count	-0.058			
Leaf area (m2)	0.999	-0.057		
DBH(cm)	0.885	-0.108	0.886	

	Water intercepted (m^3/yr)	Species count	Leaf area (m^2)	DBH (cm)
Water intercepted (m^3/yr)				
Species count	-0.057			
Leaf area (m^2)	1	-0.057		
DBH (cm)	0.886	-0.108	0.886	

Correlation is significant ($p \leq 0.00$), (n=1160)

such a pin oak, American sycamore, and red maple are among the top-performing species in this category because of their leaf structure and expansive canopies. For comparison, eastern redbud was the second most abundant species in this inventory; however, rainfall interception was the least among this species mostly because of their small canopy size. Unfortunately, similar correlations could not be shown for other environmental effects such as air pollution reduction, because the i-Tree Eco model version 6 provided summary estimates for each species rather than estimates for individual trees. However, as described by McPherson et al. (1998), However, as described by McPherson et al. (1998), pollutant uptake is a vital benefit provided by trees as they absorb atmospheric pollutants through leaf stomata and intercept particulate matter on leaf surfaces.

Recommendations

Our results show that willow oak was not ranked among the top five most abundant species, however, it ranked among the leading species for environmental effects and first for structural value. Given their estimated diameter distribution, the willow oak species has long been part of the ORNL landscape proving that it can sustain itself in a rigorous environment. However, these trees have are reaching their optimum benefit output and will likely begin to decline over the next two or three decades. Due to their size, this species has a high structural value, and so removal and placement will be expensive and ultimately lead to a significant reduction of canopy coverage. It is recommended that a mature willow tree be replaced with two medium-sized trees and three small trees to maintain current levels of canopy coverage. Willow oak is a native species to Tennessee and the southeast region, they are large, with a maximum height of 30.48 m and a maximum diameter of 101.6 cm, and provide 200-250 m^2 of canopy coverage per mature tree (USDA NRCS, 1995). This species is generally considered to be low-maintenance, rapidly growing, efficient for erosion control, and a tough tree well adapted to urban conditions. Thus it should be considered for future plantings particularly in areas that are devoid of trees on the ORNL campus.

In this assessment, Willow oak and Pin oak were identified as the leading species that contributed negatively to air pollution due to VOC emissions. There are several tree species that have a high standardized emission rate oak (*Quercus spp.*), sweetgum (*Liquidambar spp*.), sycamore (*Platanus spp.*), and poplar (*Populus spp.*). Nowak et al., (2002) suggests that VOC emissions are temperature dependent, trees generally lower air temperatures, and the addition of more leaf area/canopy cover can mitigate VOC emissions, consequently, reducing ozone levels in urban areas. Yang et al., (2015) suggests that air quality can be enhanced by strategically selecting and planting native tree species that have a higher air pollution tolerance and removal capacity in areas that are susceptible to higher pollution concentrations, such as the ORNL campus. Based on the current urban forest structure, one way to enhance the species diversity and air pollution removal capacity is to incorporate more pollution tolerant species (such as eastern white pine, eastern redcedar, bald cypress, southern magnolia) for future plantings. We suggest using more coniferous species because they have higher effectiveness for removing atmospheric pollution because of their canopy characteristics such as year-round foliage, dense and fine-textured canopies, and high leaf area index (Yang et al., 2015; Beckett *et al.*, 2000). Only two conifer species in this assessment were among the top ten more frequently occurring species. This creates an opportunity to enhance the removal of particulate matter by increasing the use of conifer species in future landscape development. We recommend decreasing the use of eastern redbud in the landscape because it occurs too frequently and despite their aesthetic value, environmental effects are not as significant compared to other species. Also, we recommend an alternative species to red maple for future plantings because it is the most abundant species and a third of its population is fairly young, indicating they are recent plantings. An alternative to these two species would be tulip poplar (*Liriodendron tulipifera*), the Tennessee state tree.

Conclusion

We hypothesized that trees in highly disturbed Block areas will have lower condition class ratings than trees in less disturbed Block areas. Our hypothesis was not supported by the results. In fact, the trees with the best condition rating were located in Block areas (B4, B5, and B6) where the most disturbance or physical activity currently takes place. An explanation for this occurrence is these are the most prominent and visible areas of the ORNL campus. Therefore, more arboriculture services are provided. Aside from this, many of the buildings are newly constructed and features new landscaping. The hypothesis that environmental effects and Importance Value will correspond more with tree size (diameter and leaf area) rather than the number of trees within a particular species was supported. Environmental effects, the dependent variable, are significantly correlated with tree size while species count was negatively correlated with the dependent variable. As previously explained Importance Value is the sum of percent population and percent leaf area, hence the species with the greatest contribution to environmental effects were among the most important species in this assessment in terms of abundance and leaf area.

The ORNL campus is relatively small in size (190.2 ha) and urban forest composition compared to most municipal forestry programs that have utilized the i-Tree Eco tool, yet it proves to be a beneficial tool for measuring environmental benefits and economic values provided by landscape vegetation. Implementing a peer-reviewed valuation model such as i-Tree Eco to estimate the structural and functional value of trees at this site helps to capture the legacy of this storied landscape, and contributes to the institution-wide commitment to research and sustainability. i-Tree Eco model estimates provide empirical evidence to answer questions related to costs and benefits that will help to guide tree-related priorities and substantiate ongoing tree management practices. While this assessment documents the current status of over 1100+ landscape trees, the ORNL Urban Forestry Program can be expanded to include unmanaged landscape, e.g. riparian areas, greenspace, other vegetative attributes, or usage of another valuation model.

Lastly, this research study sets a precedent for future urban forestry management practices at Oak Ridge National Laboratory and other government and science institution that have endured similar environmental challenges. The information derived in this research study can assist natural resource managers to inform policy, planning, and management decisions. By incorporating urban forestry management the Oak Ridge National Laboratory has the opportunity to be a leader in environmental sustainability among the Department of Energy institutions.

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PART III

ASSESSING SOIL ENVIRONS OF LANDSCAPE VEGETATION ON REMEDIATED AREAS OF OAK RIDGE NATIONAL LABORATORY

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Abstract

The Oak Ridge National Laboratory (ORNL), formally known as X-10 was established during the early 1940s to house the world's first nuclear reactor. The laboratory was used for the production and separation of plutonium during World War II, where significant amounts of chemical pollution were generated over several decades and deposited into the soil, buried, and directly discharged into local waterways. A tree assessment was conducted that coincides with a belowground assessment of landscape vegetation to determine baseline soil conditions. Soil samples were obtained from ten percent of the trees (119 out of 1160) growing on sites within the inventory. Basic soil properties, such as pH and total element content were characterized. The concentrations of twenty-one elements were determined: Al, As, Ba, Ca, Co, Cd, Cr, Cu, Fe, K, Li, Mg, Mn, Mo, Na, Ni, P, Pb, Sr, and Zn. The elemental concentrations in soils from the ORNL campus were compared to those of native soil profiles of the eastern Tennessee region and median

levels for uncontaminated world soils. A clustering analysis was used to group elements into 5 groups based on their geochemical association. A Pearson Correlation was used to determine correlations between each element which showed significant correlations between Al, Cr, Fe, K, Li, Ni, Pb, and Sr. Results show that elemental concentrations in soil samples from the ORNL site are within the ranges tabulated for soil profiles of the eastern Tennessee region, suggesting that metal contamination has not occurred.
Keywords: *Urban soils, soil contamination, trace elements, correlation analysis, and geochemical association*

Introduction

Urban soil is a general term used to describe soils that occur in urban landscapes or industrialized areas and can be identified by the presence of industrial by-products, such as (1) bricks, glass, crushed stone, industrial waste, waste incineration, garbage, processed oil products, mine spoil, and crude oil; (2) crumbled pavement, asphalt, concrete; (3) a low permeability synthetic membrane liner often used in engineering development (Rossiter, 2007). Soils in urban or urbanizing landscapes are valued for their functions to support infrastructure and roadways, regardless of potential ecosystem services (Aimone-Marsan *et al.*, 2015). Urban soils are altered by anthropogenic sources, yet they provide the same ecosystem services as natural soil systems (Effland and Pouyat, 1997). In urban landscapes, soils provide services such as retention and supply of water, carbon storage, mineral, and nutrient cycling, reduce the bioavailability of pollutants, and as a habitat for plant fauna (Bullock and Gregory, 1991; De Kimpe and Morel, 2000; Lehmann and Stahr, 2007). In urban landscapes, soil serves as the brown infrastructure of urban ecological systems, providing vital ecosystem services just as urban vegetation is considered green infrastructure in urban environments (Pouyat et al., 2007; Heidt and Neef, 2008). For example, the services provided by green infrastructure are related to vegetation and tree canopy, such as rainfall interception, mitigating urban heat island effects, and pollution removal (Akbari, 2002; Heidt and Neef, 2008). Similarly, brown infrastructure provides ecosystem services such as stormwater infiltration, detoxification, gas exchange, carbon sequestration, decomposition, and cycling of organic matter (De Kimpe and Morel, 2000; Lehmann and Stahr, 2007; Pouyat *et al.*, 2007). However, urban soils are subjected to harsher condition and lose their productive and filtering capabilities because of sealing and continuous land development.

Currently, there is a critical concern among environmental regulatory groups, urban planners, and managers regarding how to best remediate contaminated sites with effective methods to minimize environmental and social consequences. Such sites are a result of activities across various industries such as former gas stations, landfills, ammunition plants, which often lead to these sites being characterized as brownfield property in urban and rural areas where industries were present and become abandoned. The U.S. EPA defines brownfield property as "*a property in which redevelopment or reuse of it may be complicated by the presence or potential presence of a hazardous substance, pollutant, or contaminant*" (Overview of EPA's Brownfields Program, 2019). According to the U.S. EPA, there are approximately 450,000 brownfields in the United States. Furthermore, sites with extremely hazardous waste that require extensive remediation are eligible for listing on the National Priority List (NPL) and are sometimes characterized as Superfund sites by the U.S. EPA. Currently, there are over 1,300 superfund sites throughout the U.S. that must undergo a rigorous multi-phase remediation process. For example, the Oak Ridge Reservation was placed on the NPL list in 1989 as a result of activities of its three nuclear development installations. For example, during first four decades of site operations as much as 1.1 million kilograms of mercury were released into the immediate environment (Barnes, 1993), along with releases of radioactive cesium, iodine, and other radioactive products making it one of the worst environmental disasters in the Southeastern United States at the time (Bashor & Turri, 1986; Revis *et al.,* 1989). Beyond the remediation of Superfund sites or brownfield properties themselves, a priority is to contain toxic waste to prevent them from migrating to nearby communities and disrupting human health. These efforts will ensure that Superfund cleanups provide for long-term protection of human health and environmental health.

In 1942, the U.S. Army and Atomic Energy Commission (AEC) acquired nearly 60,000 of land in Oak Ridge, TN to support the Manhattan Project. The site was ideal because of its rural location but also because the Clinch River provided ample supplies of water. In addition, the Tennessee Valley Authority (TVA) could supply abundant amounts of required electricity, and there was a reliable source of labor in bordering counties and other southern cities. Beginning in 1943, there were three principal plants (X-10, Y-12, and K-25) operating in Oak Ridge, and the material produced at these facilities were used in an atomic bomb detonated in 1945. The X-10 site would become widely known as Oak Ridge National Laboratory (ORNL). Today, ORNL is a premier federal research facility that resembles a college campus and includes 1,789 hectares and 196 buildings. On an annual basis, there are approximately 5,500 users or occupants at the ORNL site (science.energy.gov). Many areas of the X-10 site have been complicated by the presence of pollutants, as nuclear development lead to a significant amount of toxic chemical waste being deposited into the soil, buried, and directly discharged into local waterways (Jean-Philippe, 2011). Environmental regulatory groups such as the DOE Oak Ridge Office of Environmental Management have committed to the remediation of contaminated soils, sediments, water, and infrastructure at the site (DOE OREMP, 2017). Remediation processes have led to the destruction of many of the facilities which ultimately has produced brownfield and greyfield areas across the ORNL site.

For several decades the Oak Ridge site has been affected by its association with environmental contamination the most commonly present are lead and polychlorinated biphenyls. "*Major operations that produced PCBs at the ORR took place from the mid-1940s into the 1970s, within the Bear Creek Valley, Upper East Fork Poplar Creek, and Bethel Valley Watersheds. Generally, contamination left the areas either as direct releases to the waterways or as indirect* *releases to soil, which then washed into the waterways and settled into the sediment*" (ASTDR, 2009). While intentional and accidental releases of radioactive and non-radioactive hazardous wastes to the immediate environs of ORNL and surrounding areas have been examined (ASTDR, 2009; Revis et al., 1989), studies conducted at the site have yet to characterize the soil environment for vegetation growing in the campus area. In an effort to remediate this site, ORNL seeks to enhance its overall environmental sustainability by incorporating urban forestry management into vegetation management. Additionally, the characterization of soil properties will help determine management practice, growth, and survival of vegetation on campus. This field study was to gather baseline data on soil composition under trees across ORNL campus. Understanding below-ground factors will help assist in developing long-term management practices for vegetation on the ORNL campus.

Methods

Site Description

Oak Ridge National Laboratory is located on the Oak Ridge Reservation (ORR) in Anderson and Roane County, Tennessee. The main ORNL site occupies approximately 1,789 hectares and includes facilities in two valleys- Bethel Valley and Melton Valley (www.energy.gov). Bethel Valley is the site of the main campus area. The site has many different land uses including nuclear reactors, research laboratories, natural areas, greyfields, brownfields, and undeveloped areas. The main campus where this soil assessment was conducted constitutes approximately 182 hectares of the total 1,789 hectares. The average low temperature is 8.4 °C, the average high temperature is 20.8°C, and annual precipitation is 139.9 cm (U.S. Climate Data, 2019). According to the Roane County Soil Survey, the general undisturbed soil types found in Roane Count are Alfisols (USDA, 1942 and 2002). The original soil in the Bethel Valley portion of ORNL was classified as either Colbert Series (fine, smectitic, thermic Vertic Hapludalfs) or Upshur Series silty clay loams (fine, mixed, superactive, mesic Typic Hapludalfs) (USDA, 1942 and 2009). The dominant forest cover type on the ORR is oak-hickory, mixed forest (pine-hardwood), or conifer (pine) (Forest Management Plan for the DOE ORR, 2015).

Sampling Scheme

A total of 1160 trees were inventoried, from there, 10 percent (119 trees) were selected for soil sampling (Figure 3.2). Three 2.5 cm soil cores, each 15.24 cm in-depth were randomly taken around each tree. At each sample site, samples were mixed, bagged and labeled. All samples were stored in a -80° freezer until analysis.

Basic soil analyses

Soil pH was determined using a glass electrode after shaking 5 grams of <2 mm soil with 10 mL of deionized water in 50 mL centrifuge tubes. Tubes were placed for 30 minutes on a reciprocating shaker at speed at 200 rev min^{-1}. Prior to measurements being taken, the pH electrode was calibrated at pH 10, pH 7, and pH 4. The H^+ activity (–log10) of each measurement was used for all statistical tests and then transformed back to pH for reporting.

Hydrofluoric aqua regia microwave digestion procedure based on Ammons *et al.* (1995) was used to measure total elemental concentrations. Each soil sample digestion was performed in duplicates. For each of the 119 samples, 200 mg of air-dried sieved soil was placed into 50 mL polyallomer centrifuge tubes, then 2 ml of reagent grade hydrofluoric acid (HF) was added to each sample and left to react for at 16 hours at room temperature. Next, 5 ml of aqua-regia (3:1:1 mixture of reagent grade hydrochloric acid (HCL), reagent grade nitric acid, and deionized water) was added to each sample and mixed with a vortex mixer. The tubes were then

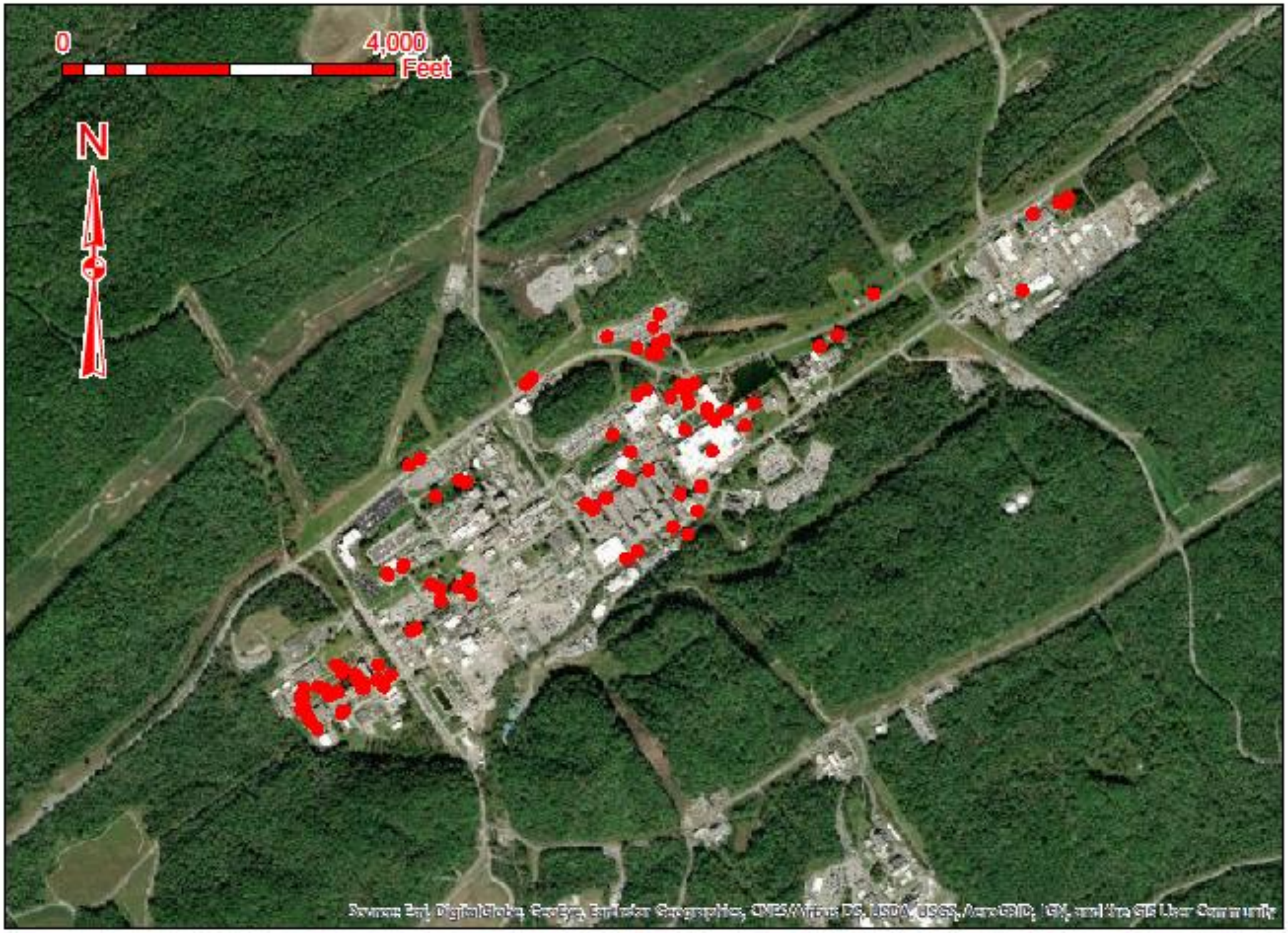

Figure 3.1. Soil sampling locations (n=119) across ORNL campus study area.

capped and placed in a Tappen 800-watt microwave oven with one beaker of deionized water present for 3 minutes at 80% power. After cooling, approximately 1 g of reagent grade boric acid was added to each sample and mixed with a vortex. Tubes were then returned to the microwave for 10 minutes at 20% power. After the tubes were cooled, they were then rinsed into 100 ml volumetric flasks with deionized water and brought to volume. The solutions were mixed and then filtered through Whatman No. 1 paper into 15 ml centrifuge tubes. The samples were then stored in a refrigerator at 4ºC until analyzed.

The extracts were analyzed using inductively coupled argon plasma-optical emission spectroscopy (ICP-OES) for 21 elements aluminum (Al), arsenic (As), barium (Ba), calcium (Ca), cadmium (Cd), cobalt (Co), chromium (Cr), copper (Cu), iron (Fe), potassium (K), lithium (Li), magnesium (Mg), manganese (Mn), molybdenum (Mo), nickel (Ni), phosphorus (P), lead (Pb), sodium (Na), sulfur (S), strontium (Sr), zinc (Zn). The method detection limit (MDL) for each element (Al (500 µg/L) , Ba (5 µg/L), Ca (100 µg/L) , Cd (10 µg/L) , Co (10 µg/L), Cr (10 µg/L), Cu (20 µg/L), Fe (50 µg/L), K (1000 µg/L), Mg (100 µg/L), Mn (10 µg/L), Mo (20 µg/L), Na (100 µg/L), Ni (20 µg/L), P (1000 µg/L), Pb (10 µg/L), S (100 µg/L), Sr (10 µg/L), Zn (10 µg/L).

Statistical Analyses

A statistical analysis using the Pearson correlation coefficient (r^2>0.5 and *p*-values <0.05) was employed to evaluate the significance of the correlation between each element. A principal components analysis (PCA) in SPSS v. 25 was used to identify patterns in data, expressing similarities and differences. JMP Pro v. 14.0.0 was used to create a Biplot to represent PCA variability and to construct a dendrogram plot to group elements and explore their association. Lastly, linear regression models were constructed in SPSS 25 to show the relationship between major and trace elements. A Student's t-test was used to assess differences in mean concertation of soils for soils that occurred along streets and roadways.

Results and Discussion

There were 21 elements analyzed in this study. However, preliminary examination of the data showed that the concentrations for the elements As, Cd, Mo, and S were below detection limits or contained insufficient data. Therefore, those elements were omitted in the statistical analyses. The elemental concentrations of 17 elements Al, Ba, Ca, Co, Cr, Cu, Fe, K, Li, Mg, Mn, Na, Ni, P, Pb, Sr, and Zn were summarized using their arithmetic mean and standard deviation values. The soil profile for the ORNL campus is then compared to background values for native soils of the eastern Tennessee region and median of uncontaminated world soils (Table 3.1). Results show that the values for ORNL soil samples either fall below or do not differ from the median levels established for eastern Tennessee soils, indicating they are within the normal range for soils in the region. When compared to background values for two watersheds in East Tennessee, ORNL soil samples were also below those values. Of the previously assessed soil, Pb wasn't reported therefore there wasn't a baseline for comparison. Additionally, the soil metal levels were below the median and within the range for uncontaminated world soils, suggesting that soil contamination has not occurred. A Student's t-test (2-tailed) was used to determine if there were differences in mean concentrations for each element for sample sites (trees) that occurred along roadways and streets. The t-test revealed that were no significant differences in mean concentrations for the elements between the street and non-street soils. The average soil pH was 6.7-7 which is optimum soil conditions for the growth and survivals of many plants.

Table 3.1 Comparison of elemental concentrations for 17 elements on the ORNL campus soils, Eastern Tennessee, and uncontaminated world soils.

Elements	ORNL	East Tennessee[a]	Oostanaula[b]	Pond[b]	Median (Range) of Uncontaminated Soils[c]
Al	4980±1833	1700-92000	52500±17700	62600±24700	71,000 (10,000-300,000)
Ba	35.9±36.5	35-570	347±143	427±149	500 (100-3,000)
Ca	834±999	63-1360	4010±6080	3850±1830	15,000 (700-500,000)
Co	1.08±0.72	<3-26	15.5±5110	18.3±6.28	8 (0.05-6.5)
Cr	4.25±1.33	<5	38.7±10.3	350±581	70 (5-1,500)
Cu	2.08±1.47	0-65	80.4±27.6	113±57.8	30 (2-250)
Fe	3479±1211	5900-74000	34600±11800	39000±17600	40,000 (2,000-550,000)
K	468±212	4000-21000	12300±4290	10900±2890	14,000 (80-37,000)
Li	2.95±1.08	-	35.0±8.70	52.9±13.6	25 (3-350)
Mg	376±206	600-7000	3600±1410	4000±1510	5,000 (400-9,000)
Mn	123±82.5	45-6000	1660±1010	1950±1110	1,000 (20-10,000)
Na	210±208	<10-3500	-	-	5,000 (400-9,000)
Ni	1.34±0.86	<10-75	-	-	50 (2-750)
P	30.2±15.3	<60-75	1140±660	1410±900	800 (35-5,300)
Pb	59.8±18.8	-	-	-	35 (2-300)
Sr	3.9±1.37	<1-47	35.6±14.3	34.1±13	250 (4-2,000)
Zn	9.60±6.92	<4-200	135±70.7	151±67.9	90 (1-900)

Elemental concentrations were determined for extracts and measured as mg kg^{-1}.

a: Data are adapted from Ammons et al., 1997
b: Data are adapted from Huangtu et al., 2019
c: Data are adapted from Bowen, 1979

The Pearson correlation coefficients (r values) for any given element pairs of these samples are summarized in Table 3.2. There were significant positive correlations (p-values <0.03 and r2> between Al (Cr, Fe, K, Li, Ni, Pb and Sr); Cr (K, Li, Ni, Pb, Sr); Sr (Ba, Ca, Cr, K, Li, Pb); Fe (Cr, K, Li, Pb); and Pb (K, Li, Ni) (Table 3.2). In this study, four principal components can be extracted accounting for 69% of the total variance. Based on the loading distribution of the variables in PCA, Al, Co, Cr, Fe, K, Li, Ni, and Pb constitute a related group (PC 1), while Ba, Ca, and Sr constitute (PC 2), Cu, Ni, P, and Zn (PC 3), and (Mg and Na PC 4) (Table 3.3). The results agree with the Pearson correlation coefficient matrix. The cumulative variability captured by the principal component axes is (PC 1: 38.6%; PCA 2: 11.8%; PCA 3: 11.0%; PCA 4: 7.5%). The data points represent the individual soil samples and the arrows illustrate the impact of each element on sample discrimination. For example, the variability of Al and Pb concentrations differentiates the samples along the PC 1 axis. For PC 2 axis, Ca, Sr, Ni, and Fe concentrations are responsible for differentiating the soils.

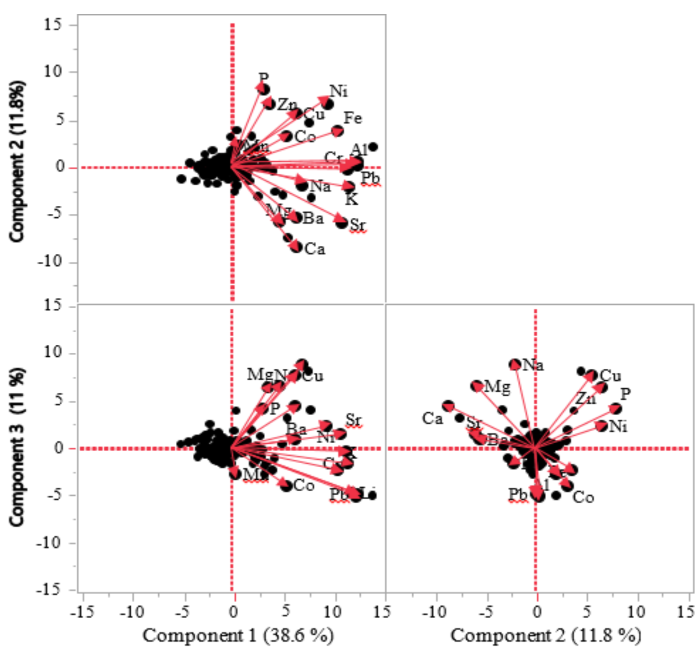

Figure 3.2. Biplot of the first three principal components and variability obtained by PCA.

Table 3.2. Pearson correlation coefficient for measured elements.

	Al	**Ba**	**Ca**	**Co**	**Cr**	**Cu**	**Fe**	**K**	**Li**	**Mg**	**Mn**	**Na**	**Ni**	**P**	**Pb**	**Sr**	**Zn**
Al																	
Ba	0.32																
Ca	0.24	0.42															
Co	0.36	0.11	-0.03														
Cr	0.69	0.32	0.28	0.27													
Cu	0.21	0.10	0.10	0.04	0.43												
Fe	0.73	0.15	0.13	0.41	0.64	0.34											
K	0.78	0.38	0.33	0.21	0.64	0.29	0.58										
Li	0.90	0.28	0.28	0.38	0.70	0.21	0.60	0.70									
Mg	0.16	0.10	0.38	0.04	0.29	0.15	0.12	0.29	0.18								
Mn	0.07	0.03	-0.01	-0.03	-0.02	0.01	0.08	-0.03	0.06	-0.16							
Na	0.22	0.21	0.38	0.09	0.42	0.47	0.30	0.33	0.20	0.59	-0.08						
Ni	0.52	0.15	0.12	0.37	0.52	0.55	0.48	0.39	0.61	0.08	0.04	0.41					
P	0.12	0.04	0.00	0.14	0.00	0.32	0.25	0.01	0.07	-0.04	0.20	0.09	0.43				
Pb	0.97	0.32	0.27	0.32	0.69	0.21	0.71	0.80	0.91	0.17	0.06	0.22	0.52	0.12			
Sr	0.60	0.61	0.79	0.18	0.54	0.21	0.42	0.68	0.60	0.32	0.05	0.40	0.34	0.15	0.62		
Zn	0.12	0.06	0.01	-0.01	0.14	0.43	0.21	0.19	0.10	0.03	-0.12	0.19	0.36	0.39	0.13	0.10	

Correlation is significant ($p < 0.03$ and $r^2 > 0.5$)

Table 3.3 Principal component loading of elemental concentrations (n=119).

Rotated Component Matrix

	Component			
	1	2	3	4
Al	.924	.249		
Ba		.713		
Ca		.851		.210
Co	.540			
Cr	.730	.247		.279
Cu			.756	.243
Fe	.773		.282	
K	.735	.390		
Li	.892	.241		
Mg		.331		.719
Mn		.216		-.626
Na		.346	.432	.632
Ni	.553		.638	
P			.737	-.366
Pb	.913	.277		
Sr	.444	.832		
Zn			.709	

This study examined geochemical associations using hierarchical clustering of elemental means were adopted from (Einax et al., 1997 and Huangfu et al., 2019). A dendrogram plot was produced to examine geochemical relationships between the elements which produced five groups (Figure 3.3). Group 1 (Al and Fe), Group 2 (Ba, P, Pb), Group 3 (Co, Ni, Cu, Li, Cr, Sr, and Zn), Group 4 (Mn and Na), and Group 5 (Ca, K, and Mg). The major elements were (Al, Fe, Ca, K, Mg, Mn, and Na) while all others were considered trace elements. Using this data, the major and trace elements that exhibited significant correlations were assessed with a linear model and the r^2 was used to determine the goodness of fit (Figure 3.4 – Figure 3.7). The observed relatively strong correlation of major and trace elements is attributed to similarities in geochemical behavior such as ionic size, ionic radius and mobility rates in the weathering environment (Nuamah, et al., 2019).

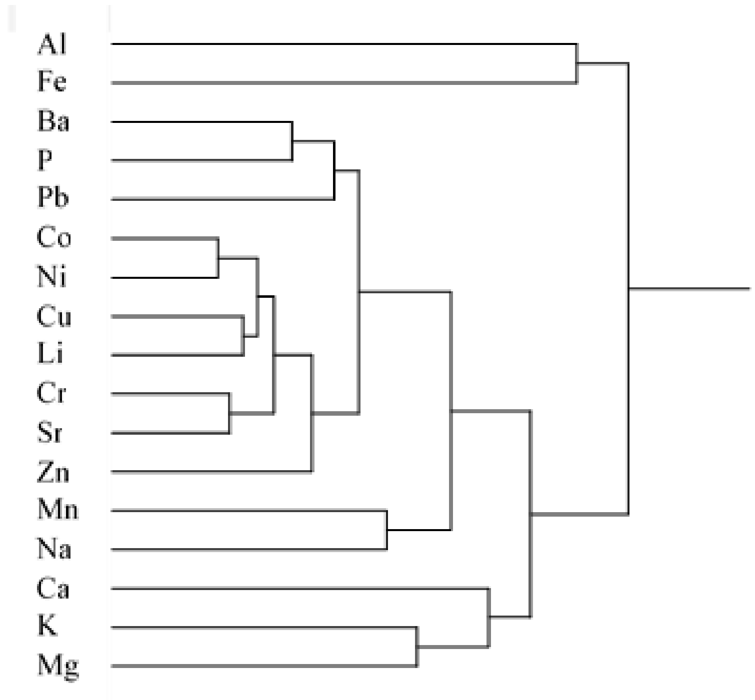

Figure 3.3. A dendrogram plot for elemental means of the 17 elements measured.

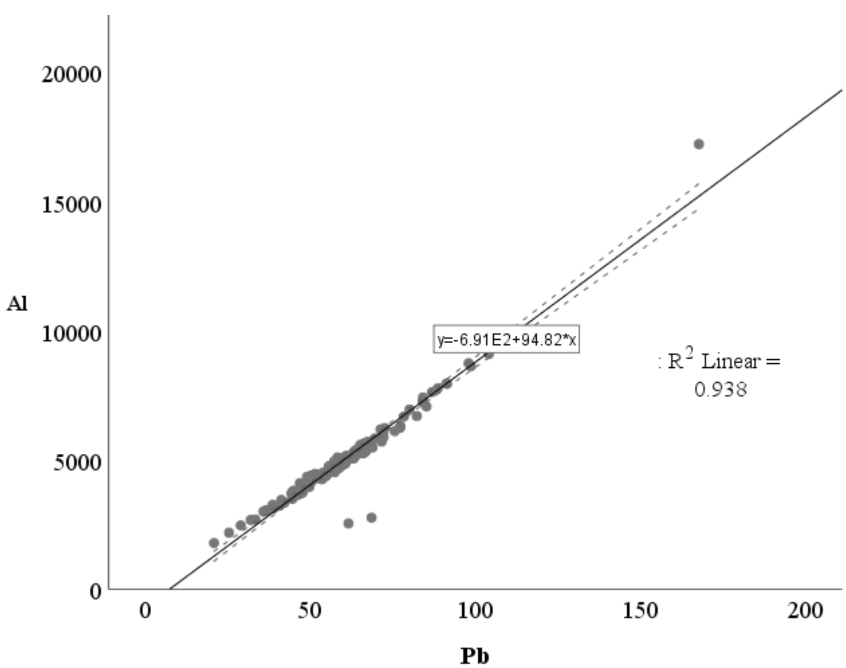

Figure 3.4. Scatter plot for linear regression model for elements Al and Pb mg kg^{-1}.

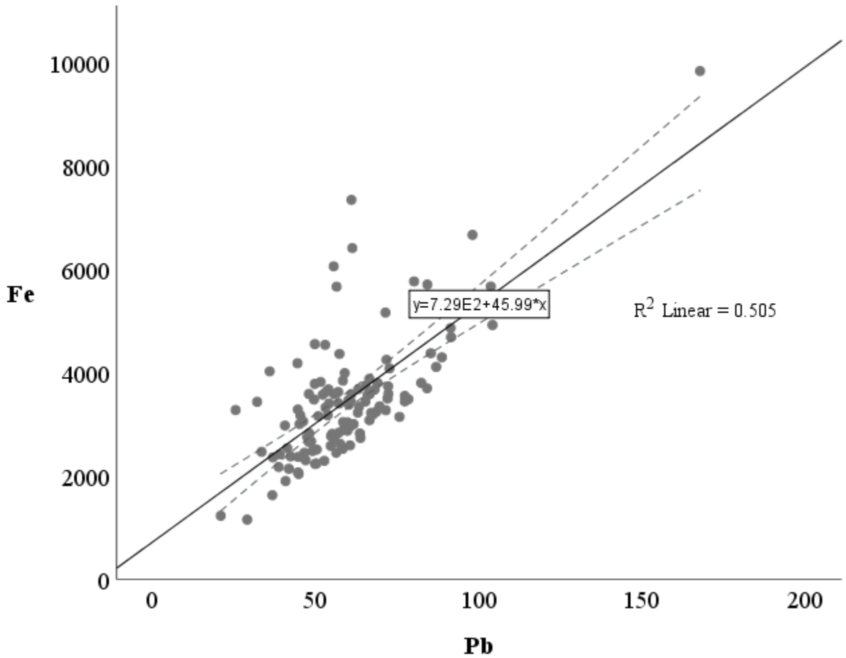

Figure 3.5. Scatter plot for linear regression model for elements Fe and Pb mg kg^{-1}.

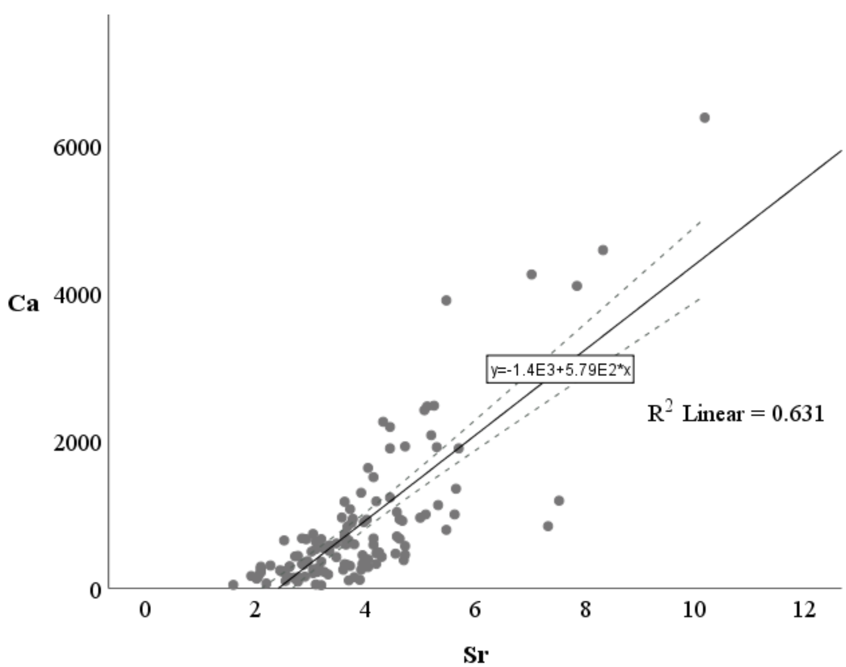

Figure 3.6. Scatter plot for linear regression model for elements Ca and Sr mg kg^{-1}.

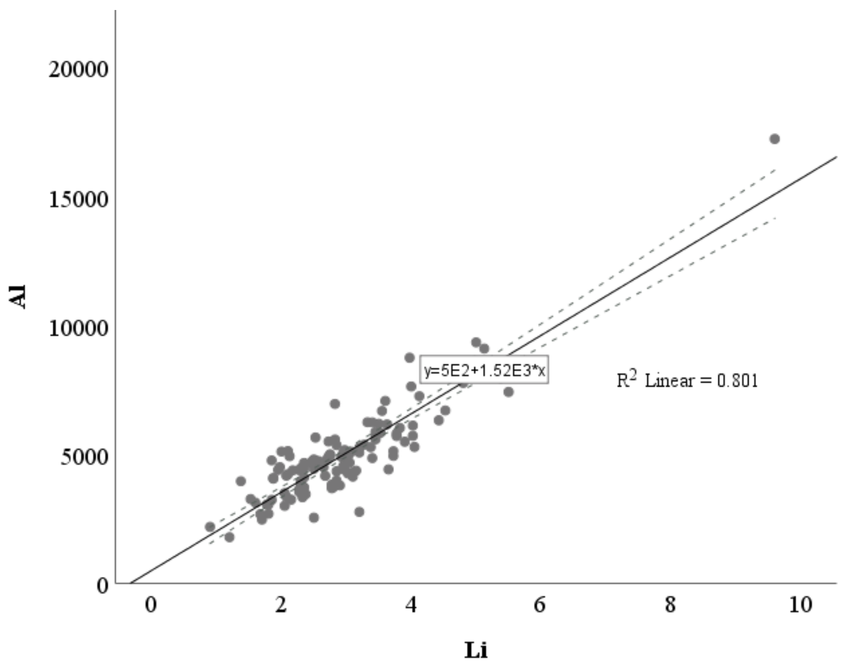

Figure 3.7. Scatter plot for linear regression model for elements Al and Li mg kg^{-1}.

Major elements such as Al, Fe, and Mn are generally linked to trace elements such as Cr, Li, and Sr, that compose the same geochemical matrix of sediments (Huangfu *et al.*, 2019; Liu et al., 2003). While the measured samples exhibited low concentrations of trace metals, this isn't a definite indicator of soil contamination. Soil contamination is referred to as soil whose chemical state deviates from the typical range but does not have an adverse effect on the environment (Knox et al., 1999). Trace element threats are not easily determined as three criteria are necessary to assess trace element threats: bioaccumulation of element, toxicity, and persistence in the soil environment (Kabata-Pendia, 2000). In addition to this, trace element bioavailability in soils is influenced by many factors such as pH, organic matter, clay, redox conditions, and total concertation (Singh, 1997; Reichman, 2002).

While the measured samples exhibited low concertation(s) of trace metals this isn't a definite indicator of soil contamination or decontamination. Soil contamination is referred to as soil whose chemical state deviates from the normal composition but does not have an adverse effect on organisms (Knox *et al.,* 1999). Trace element threats are not easily determined as three criteria are necessary to estimate trace element threats: bioaccumulation, toxicity, and persistence (Kabata-Pendia, 2000). In addition to this, trace element bioavailability in soils is influenced by many factors such as pH, organic matter, clay, redox conditions, and total concertation (Singh, 1997; Reichman, 2002).

Conclusion

The objective of this study was to document the accumulation of elements to establish background values for vegetation that constitute the ORNL campus. Additionally, this study contributes to the knowledge of soils that constitute the east Tennessee landscape. The ORNL site has a long history of environmental pollution that has impacted local soil environs, so examination of soil at this site provides insight to potential contamination. As stated in the results, trace element concertation at the sample sites did not exceed values for reference soils documented by Ammons *et al.,* (1997) nor median values tabulated for world soils (Bowen, 1997). While this study strictly assessed the chemical composition of ORNL soil profile, future studies should incorporate biological, physical, and chemical properties to assess soil quality and better understand landscape.

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PART IV

CONCLUSIONS AND IMPLICATIONS

Overview

In the early 1940s, the Atomic Energy Commission (AEC), later known as the U.S. Department of Energy designated Oak Ridge as one of three sites along with Los Alamos, New Mexico, and Hanford, Washington to design, construct and test nuclear weapons. In February 1943, construction began on the X-10 nuclear research facility, which housed the world's first nuclear reactor. The X-10 site would later become known as the Oak Ridge National Laboratory (ORNL). Today, ORNL is the largest and most diverse research development institution in the Department of Energy system. The land surrounding these sites has experienced severe physical disturbance along with inputs of toxic chemical wastes associated with its historical and contemporary use. Like any urban environment, ORNL has numerous challenges that interfere with its progress toward environmental sustainability. Natural Resource managers have worked to make ORNL environmentally sustainable through effective and strategic planning for natural resource on the ORNL campus. To guide natural resource management practice and outline goals to enhance environmental resources stakeholder developed the "Sustainable Landscape Initiative Plan 2020". One of the objectives of the Sustainable Landscape Initiative 2020 was to inventory and assess the vegetation present on the ORNL campus and quantify the environmental services associated with the vegetation (Gardner *et al.,* 2011). Natural resource managers are seeking to incorporate urban forestry management into their management practice as they work to revitalize and enhance the ORNL campus.

The goal(s) of this research project was to assess vegetation in managed landscapes and quantify the structure, function, and economic value of ecosystem services; and characterize the below-ground soil environment. I first conducted a complete tree inventory and utilized the i-Tree Eco model to investigate species diversity, species distribution, species importance, diameter distribution, tree condition, and estimate leaf area and biomass. Based on these attributes, the i-Tree Eco model estimated forest ecosystem services such as air pollution removal by species, carbon sequestration, carbon storage, oxygen production, hydrology effects, Volatile organic compound production, and monetary value of each environmental benefit. The trees inventoried have a structural value of $2.02 million, stores 29,029.9 kg of carbon and sequesters an additional 8654.5 kg of carbon per year, filters 254 kg of atmospheric pollution annually, and mitigates 273,500 gallons of stormwater runoff annually ecosystem service valued of $49,710. Secondly, I conducted a belowground assessment of landscape vegetation to determine baseline soil conditions. Soil samples were obtained from ten percent of the trees (119 out of 1160) growing on sites within the inventory and total elemental content was characterized. The concentrations of twenty-one elements were determined: Al, As, Ba, Ca, Co, Cd, Cr, Cu, Fe, K, Li, Mg, Mn, Mo, Na, Ni, P, Pb, Sr, and Zn. The elemental concentrations in soils from the ORNL campus were compared to those of native soil profiles of the eastern Tennessee region and median levels for uncontaminated world soils. Results show that elemental concentrations in soil samples from the ORNL site are within the ranges tabulated for soil profiles of the eastern Tennessee region, suggesting that metal contamination has not occurred. On average, the concentrations of Ba (35.88 mg/kg), Co (1.08 mg/kg), Cu (2.08 mg/kg), Cr (4.25 mg/kg), Fe (3479.40 mg/kg), Li (2.95 mg/kg), Mg (375.80 mg/kg), Mn (122.77 mg/kg), Pb (59.81 mg/kg), Sr (3.86 mg/kg), and Zn (9.60 mg/kg). There were strong geochemical associations between major and trace element Al, Cr, Fe, K, Li, Ni, Pb, and Sr). This information is important because background values have not been established for landscape vegetation that constitutes the ORNL urban forest.

Implications

The Oak Ridge National Laboratory is a unique institution with many natural amenities surrounding the site. In this study, the i-Tree Eco model was utilized to assess landscape vegetation and it proves to be a beneficial tool for measuring environmental benefits and economic values. Implementing a peer-reviewed valuation model such as i-Tree Eco to estimate the structural and functional value of trees at this site helps to capture the legacy of this storied landscape and contributes to the institution-wide commitment to research and sustainability. The i-Tree Eco model estimates provided empirical evidence that can inform managers of costs and benefits of urban trees. This information will help to guide tree-related priorities and substantiate ongoing tree management practices. While this study only documented the benefits of urban trees, the ORNL Urban Forestry Program can be expanded to include unmanaged landscape, e.g. riparian areas, and greenspace, therefore obtaining more insight to the ecological benefits that these natural resources provide. Furthermore, other ecosystem services assessment tools should be utilized to further examine the potential ecosystem services, benefits, and values produced by landscaped vegetation. Assessing the ORNL campus provide information about the status of trees that constate the Tennessee urban forest as this data can be used to estimate values of urban forest across the Tennessee landscape. This study also provides insight into the soil environment that constitutes the ORNL urban forest. Further study is necessary to examine other biological, chemical, and physical properties to understand soil quality therefore, management of this resource can be enhanced.

Lastly, this research study sets a precedent for future urban forestry management practices at Oak Ridge National Laboratory and other government and science institution that have endured similar environmental challenges. The information derived in this research study can assist natural resource managers to inform policy, planning, and management decisions. By incorporating urban forestry management, the Oak Ridge National Laboratory has the opportunity to be a leader in environmental sustainability among the Department of Energy institutions.

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Vita

Sally D. Ross is a native of Baton Rouge, Louisiana where she attended Southern University and A&M College and earned a Bachelor of Science in Urban Forestry in December 2015. Upon completion of her undergraduate studies, she decided to pursue an advanced degree in Forestry at the University of Tennessee, Knoxville. In fall 2016, Sally began her master's thesis work under the advisement Dr. Sharon Jean-Philippe. Sally has decided to continue her education and pursue doctoral study in Environmental Social Science at The Ohio State University.